Assessment of the Aquatic Effects of Mining in Canada:

AQUAMIN

Supporting Document II: Western Cordillera

Case Studies

103-05107/

Assessment of the Aquatic Effects of Mining in Canada: AQUAMIN

Supporting Document II: Western Cordillera

Case Studies

Prepared for:
AQUAMIN Steering Group

By:
AQUAMIN Working Group 6

1996

Working Document

Contents

Page

1 Sa Dena Hes 1
1.1 Introduction 1
1.2 Licenses, Permits, and MMLER 3
1.3 Effluent Characterization 3
1.4 Monitoring Requirements 4
1.4.1 Baseline 4
1.4.2 Operation 6
1.4.3 Closure 7
1.5 Characterization of the Receiving Environment 8
1.5.1 Response 8
1.5.2 Monitoring Adequacy 10
1.5.3 Recommendations on Monitoring 11
1.5.4 Monitoring Frequency 12
1.5.5 Special Monitoring Studies 13
1.6 Discussion 13
1.7 Conclusions 14
1.8 References 15

2 Bell Copper 17
2.1 Introduction 17
2.2 Additional Site Information. 21
2.3 Quality and Quantity of Environmental Data 22
2.4 Results of Aquatic Effects Assessment 22
2.5 Discussion 28
2.5.1 Causes and Effects Analysis 28
2.5.2 Adequacy of Monitoring Program 29
2.5.3 Applicability of MMLER and Provincial Regulatory System 29
2.6 Conclusions and Recommendations 30
2.7 References 31

3 Brenda 34
3.1 Introduction 34
3.2 Results of Aquatic Effects Assessment 38
3.3 Evidence for Aquatic Effects 42
3.4 Assessment of the Effectiveness of the MMLER and Provincial Regulatory System 43
3.5 Assessment of Monitoring-program Adequacy 44
3.6 References 44

4 Myra Falls Mine 46
4.1 Introduction 46
4.2 General Description of the Minesite 46
4.3 Assessment of Information Concerning Environmental Effects 53
4.4 Results of Aquatic Effects Assessment 56
4.5 Discussion 62
4.6 Conclusions 65
4.7 References 66

5 Cominco Snip Mine 69
5.1 Modified Case Study Assessment 69
5.2 References 74

1 Sa Dena Hes

1.1 Introduction

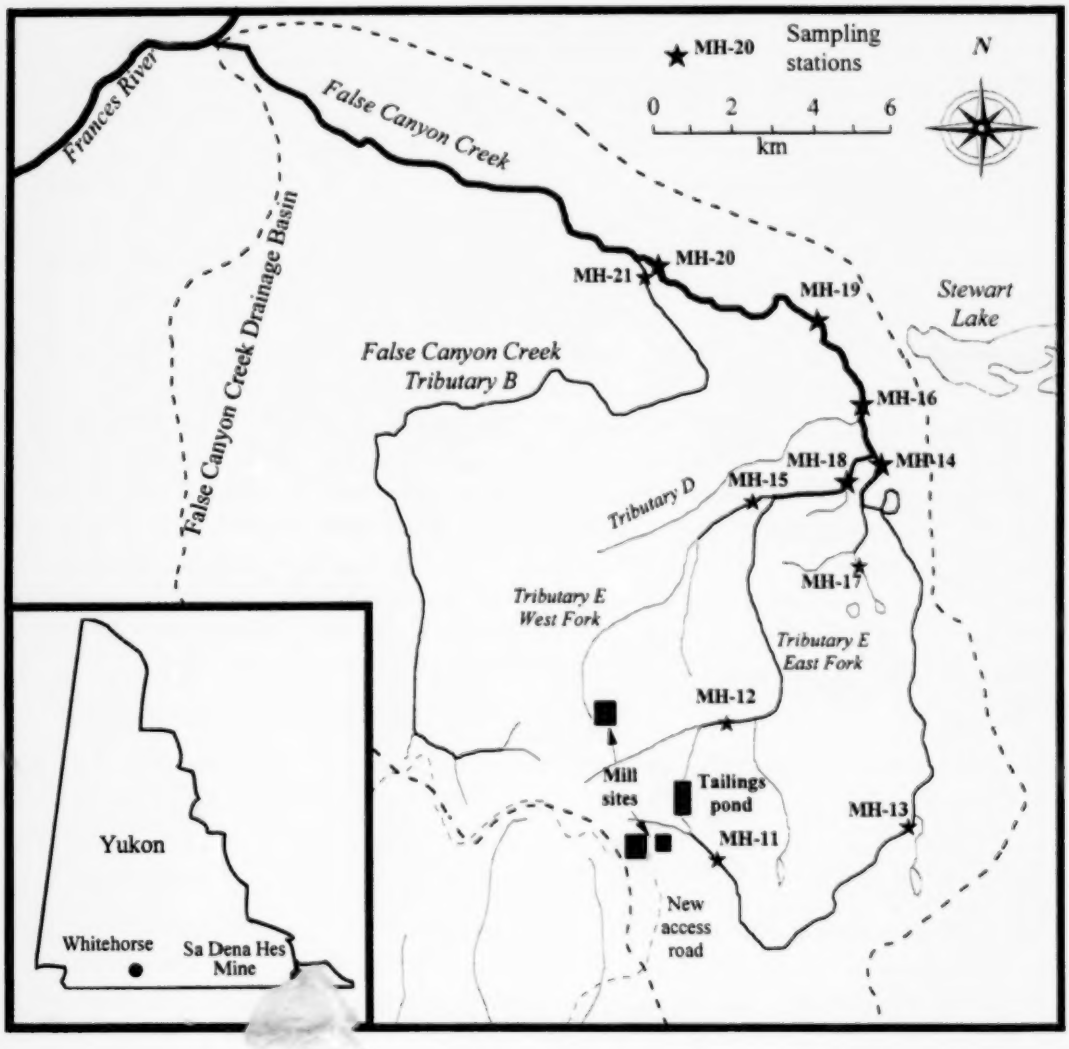
The Sa Dena Hes mine is on Mount Hundere (lat. 60° 31' 05"N, long. 128° 52' 33"W) in southern Yukon, approximately 50 km north of Watson Lake. The mine is situated on the drainage divide between the catchments of Tom Creek and False Canyon Creek (Figure 1.1). The *Yukon Territory Water Board License* stipulates that all water draining the mine workings should report only to the drainage of False Canyon Creek (Curragh Inc. 1993).

As described in the *Yukon Minfile* (1993), Zn-Pb-Ag ore at the mine occurs in several high-grade skarn zones along contacts between Lower Cambrian limestone and phyllite. Proven and probable reserves at the start of production were approximately 3.9 million tonnes in four zones, with an average grade of 3.9% Pb, 12.7% Zn, and 48 g/t Ag. The sulphide mineralization, which is fairly similar throughout the property, consists of medium- to coarse-grained sphalerite and galena more or less evenly distributed in skarn layers, with little iron sulphide present. Tailings from some of the test work have been analyzed for their acid-generation potential, and the results indicated that the tailings will be acid-consuming. Waste rock at the site is not acid-generating because the high carbonate content of the limestone host rock is coupled with the presence of insignificant amounts of pyrite and pyrrhotite.

Production from the mine, by underground and open-pit methods, commenced in July 1991 at 1500 tonnes/day, with a projected mine life of nine years. Production in 1992 was 527,000 tonnes containing 29,496 tonnes Pb, 43,445 tonnes Zn, and 29,656 kg Ag. Operations ceased on December 2, 1992, primarily as a result of low metal prices.

The ore was ground to 60% minus 200 mesh (aperture size 75 μm), and NaCN was used as a pyrite suppressant. The flotation circuit produced Pb and Zn concentrates, with Ag reporting to the Pb concentrate. Copper sulphate was added as a conditioner for the Zn circuit. From the 1500 tpd milled, 1125 tpd of tailings were produced. The latter reported to a tailings pond located at the headwaters of False Canyon Creek. Foundation seepage was expected to be high, and a reclaim pond was constructed downstream. Most of the water (87%) from the reclaim pond was recycled for mill use, but over a six-month period (mid-April to mid-October), 27% of the reclaim-pond water was released to False Canyon Creek. Year-round seepage from the reclaim pond was estimated at 27 m^3/h [Mt. Hundere, Initial Environmental Evaluation (IEE), Vol. 1 Project Overview 1990].

Figure 1.1: Sa Dena Hes mine site and sampling sites



1.2 Licenses, Permits, and MMLER

Water use for operating the Sa Dena Hes mine was governed by Yukon Water Board License IN90-002 (Yukon Territory Water Board 1991). The rate of water use was not to exceed 900,000 imperial gallons (4,091,481 L) per day, and the annual quantity was limited to 328.5 million gallons (1492 million L). In addition, water-quality criteria for waste discharge and monitoring requirements were clearly stipulated.

Of the seven parameters regulated under MMLER, the Yukon Water Board License imposed more stringent criteria for As, Cu, Pb, and Zn; the maximum authorized level for TSM was the same as that of MMLER, and Ni and ^{236}Ra were not applicable in this case (Table 1.1). Discharge limits were also set for additional parameters, including Cd, Ag, ammonia, cyanide, pH, and turbidity. The effluent standards listed in Table 1.1 were to be met at all points of entry to all receiving waters, both during active operation and after the cessation of mining.

1.3 Effluent Characterization

Except during a tailings spill on July 29, 1991, when 165 dry tonnes of tailings were discharged accidentally into the receiving environment, controlled discharge from the reclaim pond in April to October each year generally met the water-quality criteria stipulated in the Water License, and thus also for the MMLER. During spring runoff and heavy-fall rainstorms, seepage from the north tailings dam (station MH-02) and reclaim pond dam (MH-07), as well as the reclaim-pond overflow (MH-06), occasionally showed higher-than-allowable concentrations of TSM and total Pb. The flow rate, however, was usually two orders of magnitude lower than that in the two closest monitoring stations (stations 12 and 11, Figure 1.1). Consequently, dilution was expected to reduce potential impacts on the downstream water quality. For compliance record see Monitoring Adequacy.

Table 1.1: Comparison of water-license criteria (for a grab sample) and MMLER for the Sa Dena Hes mine

Parameter	Water License limit	MMLER limit (grab sample)
Ammonia (as total N)	3.50 mg/L	na
Arsenic	0.05 mg/L (dissolved)	1.0 mg/L (total)
Cadmium (total)	0.02 mg/L	na
Copper (total)	0.20 mg/L	0.6 mg/L
Cyanide (as total CN)	0.5 mg/L	na
Cyanide (as WAD CN)	0.2 mg/L	na
Lead (total)	0.20 mg/L	0.4 mg/L
Nickel (total)	na	1.0 mg/L
Radium 226 (dissolved)	na	30.0 pCi/L
Selenium (total)	0.05 mg/L	na
Silver (total)	0.10 mg/L	na
Zinc (total)	0.50 mg/L	1.0 mg/L
Total dissolved solids	50 mg/L	50 mg/L
pH	> 6.0	> 5.0
Turbidity	< 15 JTU	na
Flow	900,000 gal/day 328.5 × 10^6gal/year between April 15 and October 15	na

1.4 Monitoring Requirements

1.4.1 Baseline

On the basis of the information contained in: *i* - the Mt Hundere Project Overview, submited by Curragh Inc. in 1989, *ii* - the Regional Environmental Review Committee (RERC) guide to the IEE (1989), and *iii* - Initial Environmental Evaluation Support documents submitted in 1990, the baseline monitoring requirements were as follows:

a) Water

Water quality [field pH, dissolved oxygen, temperature, alkalinity, lab TDS, dissolved and total metals (Zn, Cu, Pb, Mn, Fe), nitrate, phosphate, sulphate] was to be monitored on a monthly basis and conducted concurrently with hydrological surveys at 13 sites, including 5 DIAND sites. The Regional Environmental Review Committee (RERC) requested at least one survey during winter low-flow, during which samples would be collected for total and dissolved metals, and analyzed by ICP scans. Groundwater samples were to be collected at one-month intervals from observation wells, and from the drainage from drift, in the fall of 1989, and were to be analyzed for the same parameters. Water levels were to be obtained from selected exploration holes. The result of the requirements was that four surveys for surface-water quality were conducted between June and September 1989 at 10 sites; total-metal ICP scans were conducted, but dissolved-metal content was determined only for Zn, Cu, Pb, Mn, and Fe.

Regarding hydrological data, monthly data were provided by Water Survey of Canada at a nearby station in operation since 1974. Nine creek gauging sites, and another one equipped with a continuous flow recorder, were visited monthly to weekly by the mine personnel.

b) Sediments

Data were not provided in the overview, and were not requested in the RERC guidelines.

c) Toxicity

There were no plans by the company for the testing of toxicity in simulated effluent, and toxicity measurements were not requested in the RERC guidelines.

d) Benthic invertebrates

Collection and archiving of 5 samples from 7 sites were to be conducted using a Surber sampler in the fall field program. Samples from the drainage of False Canyon Creek were to be analyzed to characterize the benthic community. RERC requested that analyses be done for all sites, with other samples to be archived as planned. The company actually collected triplicate samples at 23 sites and archived them, and samples from 3 selected sites were analyzed.

e) Fish

The fish study was to document the presence/absence of various life stages of fishes, to determine spatial distribution, to identify the importance of streams to those life stages in terms of rearing, spawning, or migratory corridors, to locate sensitive habitats, and to identify timing windows for acceptable in-stream construction activities and road crossings. RERC requested that fisheries studies extend beyond the area of actual physical disturbance.

1.4.2 Operation

In Yukon Water Board License IN 90-002 (Yukon Territory Water Board 1991), the following monitoring requirements are stipulated for active mining at the Sa Dena Hes mine:

a) Water quality

Eight stations within the minesite were to be monitored for field and laboratory pH, flow rate, temperature, conductivity, alkalinity, total suspended solids, dissolved solids, sulphate, hardness, total ammonia, total and dissolved Cu, Pb, and Zn, and by ICP scans for total metals. Among these, the tailings-pond outflow (station MH-1) and the reclaim-pond outflow (station MH-06) were to be monitored weekly during discharge periods from April 15 - October 15 each year, and analyses were to include total cyanide as an additional parameter. Moreover, LC_{50} bioassays were required to be conducted on water from the reclaim-pond overflow (station MH-06) on a quarterly basis.

Downstream from the minesite, 6 stations were to be monitored on a routine basis. Station MH-11 (1.5 km from the reclaim pond) was to be monitored weekly for the same parameters as were stipulated for stations within the minesite, and a scan for dissolved metals was an additional requirement for this station. Farther downstream, quarterly monitoring for field and laboratory pH, flow rate, temperature, conductivity, alkalinity, total- and dissolved-metal scans by ICP, and hardness and total cyanide were to be conducted at stations MH-12, 13, 14, 15, and 16 (see Figure 1.1 for locations).

b) Sediment quality

Commencing in the initial year of full operation and every second year afterwards, 3 replicate samples were to be collected in August at stations MH-13, 14, 18, 19, and 21 in coincidence with the benthic invertebrate monitoring program. Each sample was to be passed through a 100-mesh (150 μm) stainless-steel sieve, and the passing portion was to be analyzed for total

metals by ICP scan. During the 18-month life of the mine operation, the company sampled one set of sediments from each of stations MH-12, 13, 14, 17, 19, 20, and 21, but the analyzed samples apparently were not screened.

c) Benthic invertebrates

The sampling locations and frequency stipulated in the Water License were the same as for the sediments. Water samples were also to be collected at each site and analyzed for pH, hardness, alkalinity, sulphate, total suspended solids, total ammonia, and total and dissolved Cu, Fe, Pb, and Zn; as well, water temperature was to be measured. Prior to the mine closure, benthic invertebrates from stations MH-12, 13, 14, 20, and 21 were sampled and analyzed. The methods stipulated in the license are very general, and are subject to interpretation. No specific protocol was identified.

d) Fish

Commencing in the first year of full operation and once every two years afterwards, a fish-monitoring survey was to be conducted in September at stations MH-13, 17, 18, and 20. Prior to mine closure, a fish survey was conducted once at stations MH-12, 13, 14, 17, 19, 20, and 21, as well as at an additional site near the confluence with the Frances River (Figure 1.1).

1.4.3 Closure

Upon temporary cessation of mining, the Water License required monthly monitoring of all 9 stations within the minesite, and required water samples to be collected and analyzed for field and laboratory pH, temperature, conductivity, alkalinity, total suspended solids, dissolved solids, total metals (ICP scan), total ammonia, sulphate, hardness, and total cyanide. An LC_{50} bioassay was to be conducted on a sample from the reclaim-pond outflow (station MH-06) on a quarterly basis. Downstream from the minesite, the closest station (MH-11) was to be sampled and analyzed monthly; stations MH-13, 14, and 16 were to be sampled and analyzed quarterly for the same parameters as for stations within the minesite. Requirements for sediments, benthic invertebrates, and fish monitoring were not specified. In 1993, the water-sampling program was conducted according to the stipulations of the Water License.

1.5 Characterization of the Receiving Environment

Pre-mine baseline investigations (Steffen *et al.* 1990; Davidge and Godin 1994) indicated that the waters in Tom Creek and False Canyon Creek could be characterized as moderately hard to hard, with pH ranging from 7.5 to 8.5. The waters are clear, with episodes of turbidity during freshet. Metal concentrations were generally low in both catchments. The metals of concern, mainly Cu, Pb, and Zn, either approached or were within the same order of magnitude as CCREM (1987) water-quality guidelines for the protection of aquatic life. Because of the hardness, the CCREM maxima for Pb and Cu were set at 4 μg/L and 3 μg/L, respectively, whereas Zn remained unchanged at 30 μg/L.

The upper limit of fish distribution seemed to be 6 km downstream of the tailings-pond facility, between MH-11 (1.5 km) and MH-13 (10 km). Benthic invertebrates showed the presence of sensitive species such as *plecoptera*, *ephemeroptera*, *trichoptera*, as well as numerous chironomids. The three sites surveyed by the company were in an area that will remain relatively unaffected by the mine activities. Samples from areas that may be affected in the future are archived. Future analysis of samples representing both the baseline and the post-mining conditions will be possible only for one site, which is 24.5 km downstream of the tailings pond.

1.5.1 Response

a) Water quality

As is summarized in Table 1.2, mining activity at Mount Hundere has led to elevated concentrations of total Pb, Zn, and suspended solids in the drainage of upper False Canyon Creek. The effect is apparently limited mainly to an area that extends 2 km from the discharge point at the minesite. The high maximum values in 1991 for the three parameters (Table 1.2) are the result of the tailings spill that occurred in late July. The influence of the spill, however, did not seem to extend to10 km downstream (station MH-13). The especially low total Pb and Zn values for 1992 are probably the consequence of the addition of 5000 kg of lime to the reclaim pond in March, prior to subsequent controlled discharge in April to October.

Table 1.2: Water-quality indicators of environmental response

Variable	Year	MH-11	1.5 km d/s	MH-13	10 km d/s
		Maximum (mg/L)	Average (mg/L)	Maximum (mg/L)	Average (mg/L)
Total Pb	1991	2.5	0.1	0.011	0.006
	1992	0.646	0.068	0.034	0.014
	1993	1.21	0.117	0.04	0.017
Total Zn	1991	4.78	0.183	0.027	0.012
	1992	1.04	0.012	0.031	0.011
	1993	2.35	0.249	na	na
TSS	1991	4800	135	7	5
	1992	1540	78	30	10
	1993	1260	105	7	5

Sources: AQUAMIN documents #561 (1992 data) and #562 (1991 data) and Sa Dena Hes 1993 Annual Report to the Yukon Territory Water Board Licence IN90-002, February 1994, prepared by Coopers and Lybrand Ltd. For Cominco.

In addition to the obvious impact detected at station MH-11, subtle changes in water quality farther downstream are documented in the 1992 Annual Report (Curragh Inc. 1993). These changes include (i) at stations up to 22 km from the minesite, occasional surpassing of the CCREM guidelines for total Pb, Zn, and suspended solids during spring runoff and fall storm events; and, (ii) during effluent discharge from the reclaim pond, elevated Al, Pb, and Zn (up to 10× the concentrations in non-discharge periods) and reduced total hardness (by 30%) were apparent at all downstream stations up to 22 km from the minesite; nevertheless, only the Pb concentrations exceeded the CCREM guidelines.

b) Sediment quality

The sole sediment-monitoring program since mine start-up in 1991 was conducted in 1992 (Curragh Inc. 1993). Baseline sediment were collected and reported in Davidge and Godin (1994), however difference in station location and methodologies make the comparison difficult. Only Station MH-21 (reference site) and MH-13 (impact site) were comparable. The levels at MH-21 and station #15 (Davidge and Godin, 1994) were similar and the slight differences could be explained from the difference in methodologies since they are both reference sites. Elevated Pb was apparent in the False Canyon Creek sediments up to 22 km downstream from the minesite, and elevated Zn, As, and Cd were evident up to 10 km downstream. At station MH-14, the concentration of Pb in sediment was twice that at

reference site MH-21. At station MH-13, the Pb concentration peaked at a value four times that at MH-21. Sediment Zn also peaked at MH-13, with a concentration twice that at MH-21. The As concentration was approximately 2× at MH-11 and 1.5× at MH-13 relative to the values at reference sites MH-21 and MH-17. The Cd concentration at both MH-11 and MH-13 was about twice the background value at reference site MH-17. These numbers seem to indicate that mining at Sa Dena Hes has increased the metal content of the sediments in False Canyon Creek up to 10 km downstream from the minesite. As detailed information on the variability of replicate samples is not available, however, the accuracy of the observation cannot be ascertained without further monitoring.

c) Toxicity

The 96-h LC_{50} test performed on rainbow trout on July 30, 1991, at MH-11 (1.5 km downstream of the tailings pond) showed 100% survival. This test followed a tailings spill (165 dry tonnes) on July 29. All effluent toxicity tests (96-h LC_{50}) performed on the effluent were non-toxic (100% survival).

d) Benthic invertebrates

Most of the abundant insect populations found at stations MH-13, MH-12, MH-14, MH-20, and MH-21 were sampled.

e) Fish

A Cd content of 3.1 μg/g was obtained for the liver of graylings captured at the confluence of False Canyon Creek and the Frances River, 44 km downstream from the tailings pond.

1.5.2 Monitoring Adequacy

The *toxicity testing* on effluent in 1992 was adequate and showed the effluent not to be toxic to fish. The *water-quality testing* on effluent was adequate. The MMLER limit for Pb in grab samples was exceeded once in 4 testings, and failed the monthly mean. The limit for total suspended solids was exceeded on July 29 becaue of the spill of 165 dry tonnes of tailings. The MMLER grab-sample limit for Pb was exceeded in 1 of 14 samples of reclaim-pond seepage (MH-6). The water-quality testing permitted detection of the temporal variation in quality of the effluent discharged from the mine. However, the analytical methods for measurement of Ag (0.01 mg/l), As (0.05 mg/l), and Se (0.01 mg/l) had too high a detection

limit to allow proper assessment, particularly for downstream sites.

1.5.3 Recommendations on Monitoring

a) Water quality

Water quality downstream of the tailings pond was monitored at three principal locations (MH-11, MH-12, MH-13). Seepage reported only to MH-12, whereas seepage, effluent, and tailings (spills) reported to stations MH-11 and MH-13. The closest site, MH-11, was monitored more frequently (weekly), whereas the downstream site was monitored quarterly. Downstream station MH-13 provides an appropriate reflection of fish habitat, and MH-11 does not. The monitoring did not provide to the company a mechanism to evaluate whether problems in the downstream water quality were occurring. For station MH-11, a water-quality objective that took into account the dilution available, the background water-chemistry, and the effluent chemistry, should have been established. The availability of such numbers would have permitted the company to react earlier if some of the downstream stations could not be sampled frequently because of limited access.

b) Sediment quality

Sediment monitoring did not follow the protocol outlined in the license. The sediment was analyzed as a whole sample rather than being sieved to <150 μm as was specified. The company data for 1992 consist only of average values, and raw data are not provided. This prevents comparison with EP data for the few corresponding sites (Davidge and Godin 1994). Reference and control sites are essential for determining downstream contamination.

The Mt. Hundere mineral deposit was discovered after a 1984 geochemical assessment (Fleming *et al.* 1985) in which geochemical results were obtained for 1321 soil and stream-sediment samples collected at 50-m intervals on the property. Only four stream-sediment samples were of interest, two from each of the MH-11 and MH-12 drainage areas. The samples were sieved to <80 mesh, digested in HNO_3 and perchloric acid, and analyzed by atomic absorption spectroscopy. Significant Pb and Zn values in sediment can be found in the upper reach of False Canyon Creek (EP 1988 data, and Fleming *et al.* 1985). Temporal difference at the various sites seems to be an appropriate criterion to differentiate changes in sediment quality (provided that enough samples can be collected to make statistically valid comparisons).

The utilization of different locations as a substitue for the lack of temporal data creates its own difficulties. Doherty *et al.* (1994) show the lower reaches of False Canyon Creek to be

glaciolacustrine sediments. The unknown provenance of these sediments makes them unsuitable as a reference for metal levels for the minesite.

c) Benthic invertebrates

Several methodology problems were encountered in the studies of benthic invertebrates. The use of different mesh sizes, sample sizes, and collection times between those of the baseline study and the impact study make it difficult to interpret the information. As well, control sites were not adequate. Some attempts were made to standardize the monitoring of benthos information (Environment Canada 1993).

Interpretation of data for benthic invertebrates was confounded by changes in the physical characteristics of the stream banks, and natural variability. These effects were additional to the introduction of metals into the receiving environment. The magnitude and effect of these factors on the changes in water-quality response are not known. Better control of variables may require different approaches.

d) Fish

Metal contents of fish tissues are difficult to interpret. No substantial reference on levels of metals in tissues is available. The survey did not benefit from baseline studies to establish pre-production metal concentrations.

1.5.4 Monitoring Frequency

a) Water-quality data for the MH-13 site were collected too infrequently to determine whether CCREM water-quality limits were significiantly exceeded. To enhance protection, sampling frequency should be increased if problems are suspected (*e.g.*, at MH-11). The lack of appropriate frequency in sampling prevents the determination of statistical differences between the baseline and subsequent information. For example, the information contained in Steffen *et al.* (1990) and Davidge and Godin (1994) for Pb showed powers of 72% in 1991, 29% in 1992, and 19% in 1993 (using alpha = 0.05).

b) The frequency of sediment-sampling should be biannual, or at least annual. Biannual sampling would correspond with major hydrological events such as summer low-flow after freshet, and winter low-flow after the autumn rains.

1.5.5 Special Monitoring Studies

a) Acute toxicity testing in the receiving environment is not appropriate. The data collected by the company at least provided good evidence to show that the tailings spill was not toxic. Information for the receiving environment, however, should be directed towards sublethal and chronic effects rather than acute responses.

1.6 Discussion

As the Sa Dena Hes Mine operated for only 18 months, relatively few monitoring data have been collected to date to allow a comprehensive assessment of the impacts of the mine effluent on the receiving environment. The existing data set, however, still sheds light on three relevant issues:

1) Effectiveness of MMLER

The fact that the Yukon Water Board License (IN90-002) imposed limits more stringent than MMLER with respect to the quality of the mine effluent from Sa Dena Hes indicates that the local authority considered MMLER insufficient to protect the aquatic environment downstream of the discharges from the mine. However, to the extent that the revised criteria were largely based on MMLER, the latter remain a useful regulatory framework. Discharges from the mine met the specific Water Board License effluent-quality criteria most of the time during the mining operation, and no acute deleterious effect on the biological community downstream was apparent. However, the limited environmental monitoring data-set does not address chronic effects, nor does it allow an estimation of potential acute effects if the MMLER were to be used.

2) Adequacy of the monitoring program at Sa Dena Hes

With the emphasis of monitoring largely placed on water quality within and near the Sa Dena Hes minesite, the available monitoring data-set contributes little to detect changes, other than in water chemistry, in the aquatic resources downstream. This deficiency is in part the result of the short lifespan of the mine and the infrequent sampling of downstream locations as required by the Water Board License. The frequent omission of sampling of a reference site, the lack of baseline information, and inconsistencies in sampling protocols make comparative assessments particularly difficult.

3) Requirements for a prospective environmental effects monitoring (EEM) program

On the basis of the information contained in the reviewed reports and summarized above, it is evident that the monitoring scheme applied at the Sa Dena Hes mine has been largely deficient in providing the data necessary for environmental effects assessment. To improve the situation, at least five issues should be addressed: (i) more frequent sampling of downstream monitoring sites (yearly instead of every other year); water-quality objectives could be derived if access to fisheries habitat is difficult; (ii) sampling standardization, in that water, sediment, benthos, and fish should be collected at the same sites; (iii) inclusion of reference sites, both spatial and temporal; (iv) establishment of sedimentation rates at downstream sites (because water frequently contains excessive TSM), and determination of the bioavailability of the contained metals; and (v) use of more sensitive analytical techniques to measure Ag, As, and Se in order to facilitate assessment of chronic effects on the biota.

For metal mines in general, a well-designed program for monitoring environmental effects should include a proper choice of sampling stations, adequate sampling frequency, use of appropriate analytical techniques to measure specific parameters, consideration of chronic as well as acute effects, and determination of the fate and transport of the metals of interest.

1.7 Conclusions

On the basis of a critical review of six reports on the Sa Dena Hes mine, the following can be concluded:

1) During the short period of mining at Sa Dena Hes, a temporary increase in TSM, and total Pb and total Zn, and a decrease in total alkalinity were detected in False Canyon Creek up to 22 km downstream from the minesite.

2) An increase in Pb in sediments was apparent 20 km downstream, and Zn, Cd, As, and Cd increased 10 km downstream of the minesite. In the absence of a detailed characterization of the reference sites and lack of information on variability of replicate samples, however, the accuracy of the observations cannot be ascertained.

3) As the Yukon Water Board License criteria governing the effluent quality from the Sa Dena Hes mine are more stringent than those of the MMLER, the limited data-set acquired to date at the mine cannot be used to assess the effectiveness of MMLER in protecting the downstream aquatic resources along False Canyon Creek.

On the basis of the Sa Dena Hes experience, it is recommended that an environmental effects monitoring program for metal mines should include careful consideration of a proper choice of monitoring sites, the use of adequate sampling frequency and appropriate analytical techniques, the bio-availability of metals, and both the chronic effects and the acute effects on biota.

1.8 References

Canadian Council of Resource and Environment Ministers (CCREM) (1987) *Canadian Water Quality Guidelines.*

Curragh Inc. (1989) *Mt. Hundere - Project Overview and Plan for Initial Environmental Evaluation.*

*Curragh Inc. (1993) *Sa Dena Hes - 1992 Annual Report*. Unpublished report prepared for Yukon Water Board, 222 p. (AQUAMIN Report 561).

*Davidge, D. and Godin, B. (1994) Mt. Hundere baseline study - June 1988 and June 1990. *Environment Canada Environmental Protection Service, Pacific and Yukon Region, Regional Program Report (Draft)*, 18 p. plus appendices. (AQUAMIN Report 566).

Doherty, R.A., Mougeot, C.M. and Vanranden, J.A. (1994) *Geological Processes Inventory Map of Watson Lake 105 A - Terrain Hazards, Geological Processes, Faults, Seismic Events.* Indian and Northern Affairs, Whitehorse, Yukon.

Environment Canada (1993) *Guidelines for Monitoring Benthos in Freshwater Environments*. North Vancouver.

Fleming, D.B., Hodgson, C.J. and Hitchins, A.C. (1985) *1984 Geochemical Assessment Report - Mt. Hundere Property*. Canamax Resources Inc.

Regional Environmental Review Committee (RERC) (1989) Letter from Department of Indian and Northern Affairs to Curragh Inc. on Mt. Hundere Initial Environmental Evaluation.

*Steffen, Robertson and Kirsten (B.C.) Inc. (1990) *Mt. Hundere Initial Environmental Evaluation (Biophysical Evaluation) Volume IV*. Unpublished report prepared for Mount Hundere Joint Venture. (AQUAMIN Report 564).

*Yukon Territory Water Board (1991) *Yukon Water Board, IN90-002 - Mt. Hundere Water License.* Water-use license and conditions, issued to Mount Hundere Joint Venture, Whitehorse, Yukon. (AQUAMIN Report 562).

AQUAMIN Reports

561: Curragh Inc. (1993)
562: Yukon Territory Water Board (1991)
564: Steffen *et al.* (1990)
566: Davidge & Godin (1994)

2 Bell Copper

2.1 Introduction

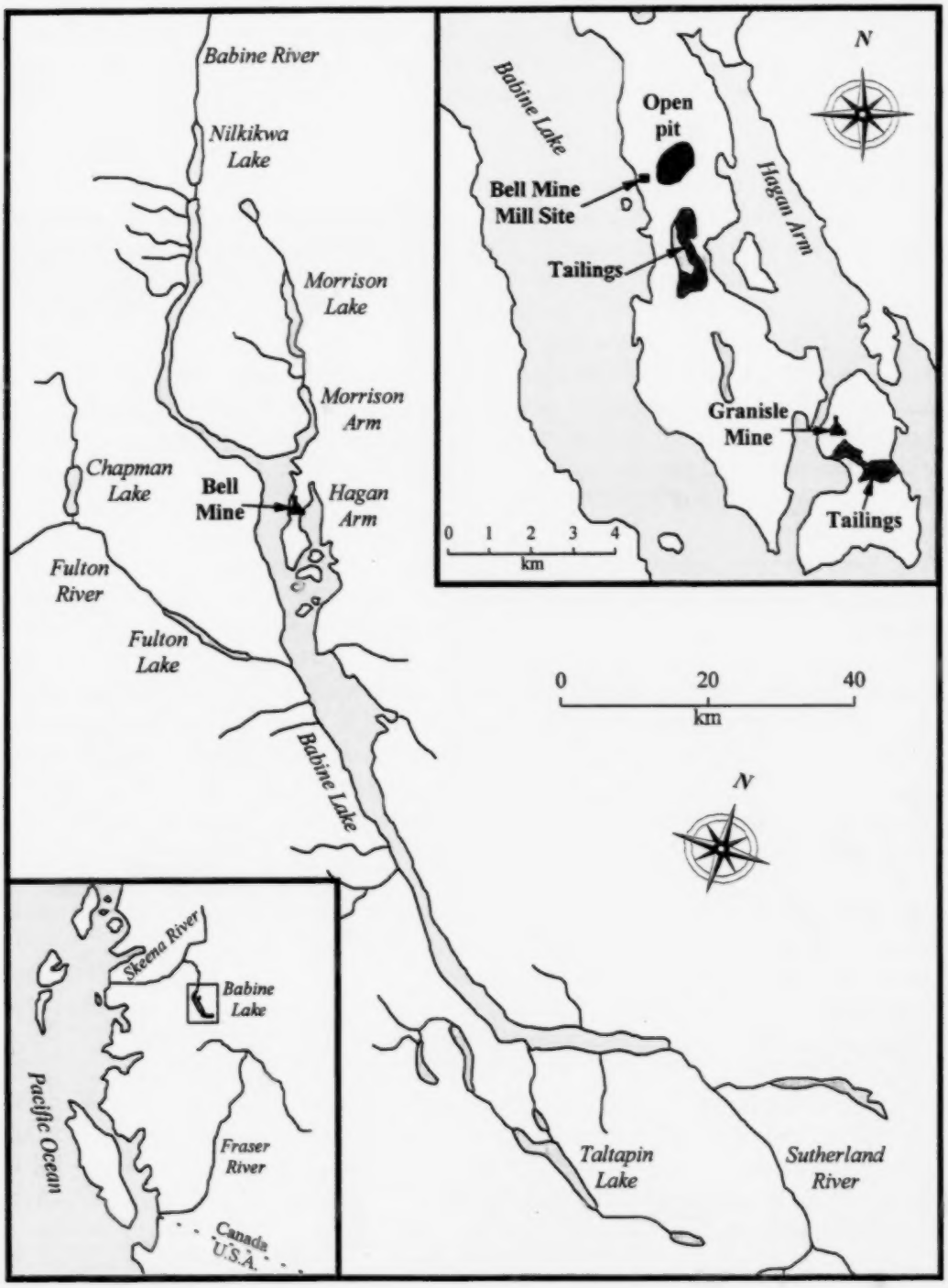
The Bell Copper mine, which ceased production in 1992, is at lat. 54^0 59'N and long. 125^0 14'W on Newman Peninsula in Babine Lake, approximately 65 km east north east of Smithers, British Columbia (Figure 2.1). The local geology of the Bell Copper mine has been described by Carson *et al.* (1976). The mineralization is typical of the calc-alkaline suite of porphyry copper deposits of the Canadian Cordillera. The orebody, 150-300 m wide by 1000 m long, follows and overlaps the western and northern edges of an Eocene plug of biotite-hornblende-plagioclase porphyry. This plug intruded Jurassic and Cretaceous volcanic and sedimentary rocks and Eocene rhyodacite. Chalcopyrite and minor bornite are the main ore minerals, and abundant pyrite occurs in a peripheral halo 300 m wide. Prior to the commencement of mining, the geological reserves were 116 M tonnes averaging 0.48% Cu, about 0.35 ppm Au, and <0.005% Mo.

The open-pit mining operation started in 1970 with overburden and cap-rock stripping and the construction of mining facilities. Milling commenced in 1972, with an average ore-processing rate of 9100 tonnes per day (tpd) (10,100 tons per day) gradually increasing to 15, 500 tpd by 1981. Milling operations ceased in October 1982 due to deteriorating mineral markets. During temporary closure, limited prestripping occurred between 1983 and 1985 when the mine reopened at a milling rate of 15, 500 tpd. It was proposed to extend the mine life to 1993, however, due to difficult mining conditions and low copper prices, certain ores were considered uneconomical and mining operations were terminated in March 1992 when available ore was exhausted (Noranda Minerals, 1994). Copper was concentrated by the flotation process at a mill on site.

The Bell Copper facility occupies approximately 550 hectares, with the major disturbances (rock piles, tailings pond, etc.) covering 419 hectares (Carson *et al.*, 1976). The open pit measures 670m by 914m and is 290m in depth. Over the life of the mine, pit production totalled 137M tonnes mined including 71M tonnes of ore milled and 66M tonnes of mine development rock removed (Noranda Minerals, 1994).

The tailings impoundment, containing over 57M m^3 of tailings (Noranda Minerals, 1994) is located in a basin formerly occupied by Workburn Lake. Impoundment construction utilized mine rock, with cycloned tailings deposited on the upstream face to aid in sealing the voids in the rock mass. Consequently, seepage ponds are required to collect the water which passes through the dam structures. The seepage waters were returned to the tailings impoundment

Figure 2.1: Location and general setting of the Bell Copper mine.



during the operating life of the mine.

Acid rock drainage (ARD) has been associated with the oxidized cap-rock, which was incorporated into rock piles and tailings dams. Typical ARD at the Bell mine has the following water quality characteristics: 0.1-10.0 mg/L Cu(d) and 0.05-0.50 mg/L Zn (d), with a pH of 6.00 (Noranda Minerals, 1992b). However, a minesite runoff sample with a pH of 4.0 and containing as much as 21.1 mg/L Cu, 16.5 mg/L Zn, and 12.6 mg/L Al was collected in 1984 (Godin *et al.* 1985). After reopening of the mine in 1985, more effective seepage-control measures were initiated to reduce high-metal runoff. Efforts are made to collect and pump ARD waters to the tailings impoundment, where a high pH is maintained (Noranda Minerals, 1992).

Prior to the 1982 shutdown, Bell Copper operated with a closed water circuit, thus eliminating the need for direct discharge of mine effluent into Babine Lake. During the temporary shutdown it was necessary to prevent uncontrolled overflow from the tailings pond in order to avoid structural damage to the retainment dam. The discharge was also necessary to maintain the open pit in a relatively dry state for possible resumption of mining operations. To this end, in October 1982 the mine was granted limited approval by the provincial government to discharge up to 1 M m^3 of tailings-pond supernatant to Hagan Arm over the period October 1982 to April 30, 1983 (B.C. M.O.E. 1982). This approval was subsequently amended to allow the discharge to continue to the end of July 31, 1983. Under the Letter of Approval the mine intermittently discharged the following volumes of supernatant to Hagan Arm:

- Oct. 1982 -January 25, 1983: 729,108 m^3; average 0.032 mg/L dissolved Cu (Basu 1984)
- May 12, 1983 - June 1, 1983 : 144, 056 m^3; average 0.033 mg/L dissolved Cu (Kennedy 1983a)
- June 21, 1983 - July 4, 1983 : 116, 922 m^3; average 0.032 mg/L dissolved Cu (Kennedy 1983b)

Between October 1982 and July 4, 1983, a total of 1 M m^3 of supernatant with an average concentration of 0.032 mg dissolved Cu/L were discharged into Hagan Arm. A May 10, 1983, bioassay conducted on a sample of the supernatant showed the discharge to be non-toxic (no mortalities of test fish over 96H using 100% effluent; Kennedy 1983a).

In October 1983 the provincial waste-discharge permit (# PE-1505) was amended to reflect a change from a total recycle operation, allowing the mine to discharge supernatant intermittently from the tailings pond to Babine Lake via a submerged outfall at a depth of at least 5 m below the water surface (B.C. MOE 1983). The permit allowed the mine to discharge a maximum of 9000 m^3/day (maximum of 1 M m^3 annually during years of higher

rainfall). The maximum concentrations of dissolved metals allowed were 0.05 mg/L Cu, and 0.2 mg/L Zn, and the discharge was to be non-toxic (96-h LC50 >100%). The permit was amended in November 1984 to, in part, remove the "during years of high rainfall" limitation. (B.C. MOE, 1984).

Environment Canada (Environmental Protection Service) site monitoring in May 1983 and July 1984 showed that only one supernatant sample (total of eight samples) with a dissolved copper content of 0.112 mg/L) was out of compliance with the Bell Mine waste discharge permit (#PE-1505) maximum allowable level of 0.05 mg/L (Godin *et al.* 1985). A further sample taken later the same day was within permit limits for copper (i.e. 0.034 mg/L dissolved Cu).

Under direction of its provincial waste discharge permit, the Bell mine also conducted frequent monitoring of tailings pond supernatant to ensure adequate quality for discharge to Babine Lake. A cursory review of Annual Envrionmental Reports indicates that copper levels were normally within permit limits and the effluent was non-toxic (ie. 100% survival over 96H) to test fish. Following is a brief synopsis of discharge information for the period 1989 to 1993. Additionally all bioassays (4) conducted in 1985 (discharge of 342, 960 m^3) showed the tailings pond supernatant to be non-acutely lethal to rainbow trout (96H LC_{50} = 100%). (Noranda Minerals 1985 to 1994).

Synopsis of Supernatant Discharge to Hagan Arm (Rum Bay) (1989 to 1993):

	1993	1992	1991	1990	1989
Volume (1000 m^3)	928	947	649	429	479
Cu (d) mg/L (Range)	0.019 (0.004 - 0.039)	0.013 (0.002-0.041)	0.016 (0.001-0.049)	0.008 (0.002-0.020)	0.013 (0.004-0.110*)
Bioassay (LC_{50}) (# Tests)	100% (1)	100% (5)	100% (4)	100% (2)	100% (5)

* Supernatant was not discharged during the maximum copper (0.110 mg/L) sampling event.

In addition to the intermittent discharge allowed during the tempory closure (October 1982 through July 1983) and the ongoing, intermittent discharge authorized under the amended permit, there have been a number of effluent spills from the mine (B.C. MOE 1989). One of the more significant events occurred on December 12, 1989, when approximately 4.5M litres of tailings-pond waters spilled through the tailings-dam structure into a seepage pond and thence overland into Rum Bay (Hagan Arm). A bioassay taken during the spill incident showed the effluent to be non-toxic; Cu concentrations ranged between 0.308 and 0.015

mg/L at that time.

2.2 Additional Site Information

Babine Lake, with a surface area of about 490 km^2 and a drainage area of about 10,000 km^2, is the largest natural lake in British Columbia. The lake is one of the major producers of sockeye salmon in the province, accounting for about 90% of the sockeye salmon production from the Skeena River system. Up to 1.5 million adult sockeye, pink coho, chinook, and chum salmon return annually to spawn in the system. Construction of sockeye salmon spawning channels on the Fulton River and Pinkut Creek has led to a three- to four-fold increase of sockeye fry loading over historical levels, and up to 200 million sockeye salmon juveniles now rear in the system. Besides salmon, resident fish include Kokanee, Rainbow Trout, Lake Trout, and Burbot, which are popular for sport fishing.

The Granisle mine operation, located immediately south of the Bell Copper mine (Figure 2.1) operated from 1966 to mid-1982. The mine embayed several areas of Babine Lake for tailings disposal. Although the impoundment dikes were constructed of relatively coarse material, they were sealed by cycloning tailings on the inside face and the mine recycled waters to the mill. Accordingly, while it is possible that some metals may have been released to the lake as a result of seepage it is expected that this contribution would be relatively minor. Preliminary studies conducted in the early 1970's found that the lake waters immediately adjacent to the impoundment area were not significantly elevated with metals (*i.e.* copper). However, at the time there was evidence of tailings on the lake bottom as far as 30 metres from the tailings dam. These deposits were believed to result from tailings spilled during construction of the No. 2 dam (Kussat *et al.*, 1972). Other incidental discharges from the site would similarly have contributed metals and other contaminants to the lake. One such example, was a seep discovered draining into Babine Lake from the toe of the Granisle mine North Dump (north end of McDonald Island - Environment Canada, 1990). Although a bioassay sample revealed the seepage to be toxic to fish (100% mortality of test fish in <24H) the extent of the impact on the lake was not specifically determined. The seep was secured and all seepage collected and stored in the Granisle pit (McArthur, 1995). As in the case of the Bell mine the Granisle mine had a number of tailings spills during its operating life. The presence of the Granisle mine must therefore be considered when interpreting data from Bell mine studies, particularly for those sites located in the vicinity of the former Granisle operation.

In addition to mining and fish propagation, there is also considerable active logging in areas adjacent to the Bell minesite and elsewhere throughout the Babine Lake watershed. These operations could also potentially have effects on the Babine Lake ecosystem.

2.3 Quality and Quantity of Environmental Data

For the Bell Copper case study, the assessment of aquatic effects is based on a critical review of an Environment Canada regional data report (Godin *et al.* 1992) and the following four reports in the AQUAMIN Database: 124 (Basu 1984), 273 (Rescan 1992a), 274 (Rescan 1992b), and 545 (Godin *et al.* 1995). These reports review data mostly from chemical and biological surveys conducted in a 21-year period between 1971 and 1992. The data-set covers both historical and existing environmental conditions of Babine Lake as a whole. The emphasis of the present assessment, however, is on the conditions of the immediate environs of the Bell Copper mine. A cursory review was also made of annual mine environmental monitoring reports for the period 1985 to 1993; excluding those for 1987 and 1988).

The environmental data contained in the five reports were originally acquired using well-documented and accepted methodology at the time of collection. Most of the reports are believed to reflect accurately the environmental conditions of the locales from which the data were derived. However, many of of the investigations and surveys were designed for specific and often diverse purposes e.g., the report by Godin *et al.* (1985) was in response to a short-term discharge of tailings supernatant into Hagan Arm, and that by Rescan (1992b) was for the development of a closure plan. Therefore, *in toto* the data lacks the consistency of a comprehensive and integrated environmental monitoring program for the Bell Mine site. Consequently, the data do not permit conclusive cause-and-effect analysis for all of the environmental parameters considered important in the AQUAMIN exercise. For example, information for the pre-mine and early mining periods is somewhat lacking, and insufficient data have been gathered to allow statistically sound, long-term evaluations of biological parameters to be conducted. These observations will be discussed further in subsequent sections of this case study.

2.4 Results of Aquatic Effects Assessment

a) Water Quality

Babine Lake is dystrophic (i.e., rich in humic substances) and oligotrophic (i.e., poor in nutrients, especially phosphorus). The lake is characterized as slightly basic (pH uniformly about 7.7), with low total alkalinity (approximately 37 mg $CaCO_3$/L). Given the large size of Babine Lake and the relatively small volumes of generally good-quality effluent discharged, it is not expected that the mining activity would have a significant effect on the overall quality of the lake water. The extensive water-quality database for the main body of the lake has confirmed the relatively stable nature of lake-water chemistry since 1974 (Rescan 1992a,b). However, the water chemistry of Hagan Arm, which received the bulk of mine effluent since 1982, differs markedly from that of the main body of Babine Lake,

particularly in dissolved Cu (Figure 2.2), sulphate, chloride, and to a lesser extent, hardness and Zn (Rescan 1992a,b; Godin *et al.* 1985).

Figure 2.2 includes data from Godin *et al.* (1992), but a limited number of results from the early 1970s for Hagan Arm (reviewed in Rescan 1992b) are excluded because of the high detection limits for these earlier analyses. Nevertheless, these earlier data suggest that, prior to the start-up of the Bell mine, the water chemistry of Hagan Arm was very similar to that of the Main Arm of Babine Lake. Possibly the apparent reduction in Cu concentration in Hagan Arm since the late 1980s is attributable to the installation of more efficient seepage-collection systems at the Bell mine and control of effluent quality prior to discharge into Hagan Arm (e.g. frequent monitoring of supernatant quality, discharge only when quality and quantity were within permit limits). The recent improvement in water quality in Hagan Arm also indicates that the effect of the mining operation on the water chemistry of the receiving body is only temporary.

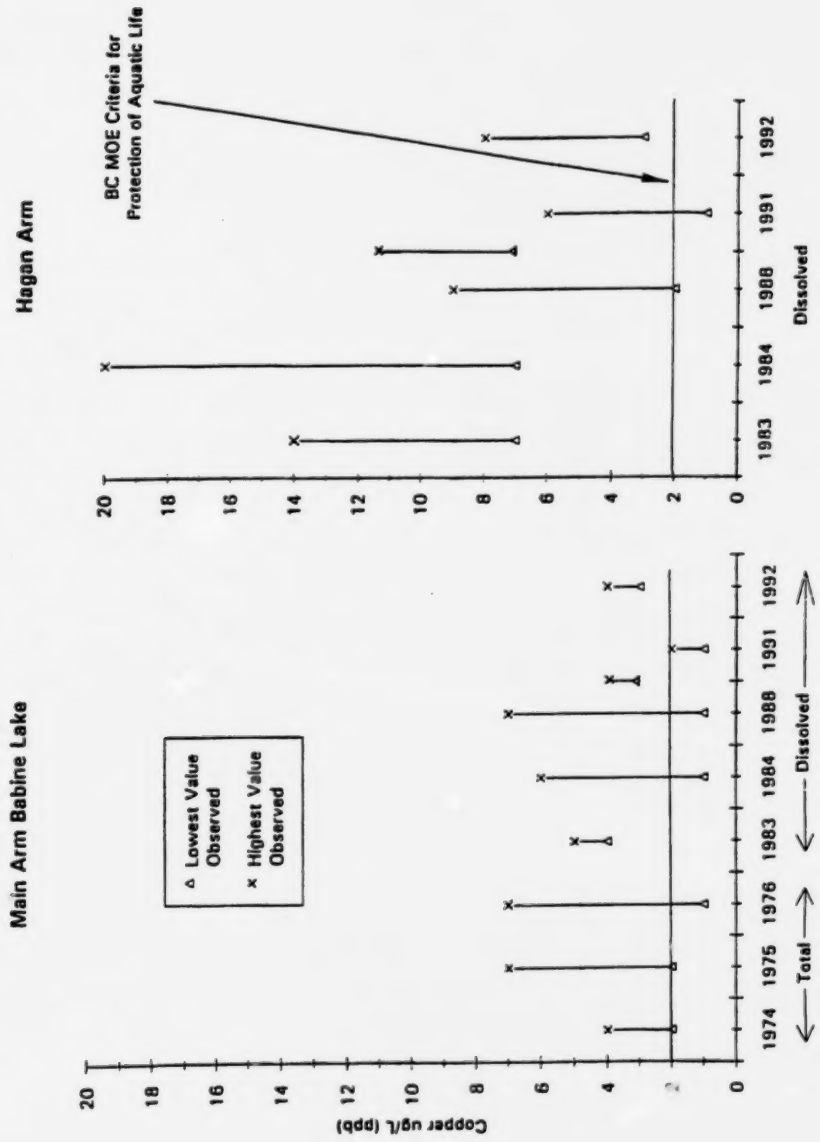
It has been noted that water in Babine Lake, including Hagan Arm, has a high organic complexing capacity for Cu (Rescan 1992a,b). This may reduce potential adverse impacts on the biota even though the background dissolved-Cu content commonly exceeds the BC MOE criteria for protection of aquatic life (Figure 2.2).

b) Sediment Quality

Relative to a background value of 30-100 ppm Cu, the top layer (<10 cm) of near-shore sediments in Newman Bay, Rum Bay, and Woolverton Bay (all in the vicinity of the Bell mine; Figure 2.1) exhibit a higher Cu concentration by as much as two orders of magnitude. There is some evidence (petrographic and sequential extraction analyses) that the bulk of Cu in the Newman Bay and Rum Bay sediments may occur in solid phases (e.g., chalcopyrite) that will not be easily mobilized (Rescan 1992b; Godin *et al.* 1992). As recorded in Rescan (1992b), the analysis of two cores from Woolverton Bay indicates that elevated Cu concentrations are limited to the top 5 cm of the sediments; the near-shore core has a higher Cu content (9000 ppm) than the deeper core (3000 ppm). No sequential extraction analysis, however, has been conducted to clarify the nature and geochemical behavior of the constituent Cu-bearing phases. Dissolved Cu in the pore water at the sediment water interface at both core sites is at least twice that in the water column, indicating diffusion of Cu into the overlying water column. The net flux is apparently small, potentially contributing only 0.6% of the Cu concentration commonly measured in Hagan Arm.

Godin *et al.* (1992) also found that Cu tended to be much higher in the top centimetre of sediment. They found maximum sediment copper values in surface layers at a sampling site located at the extreme north end of McDonald Island. However, given the distance from the

Figure 2.2: Copper concentration observed in Babine Lake and Hagan Arm (after Rescan 1992 (b) with added 1990 data from Godin *et al* 1992)



Bell Mine (Figure 2.2) the site may be more influenced by other sources of metals, such as he Granisle Mine, although the latter was not the subject of extensive review in this excercise.

c) Toxicity

Since 1978, the Bell mine has conducted routine acute-toxicity tests (96-h LC_{50}) on rainbow trout exposed to tailings supernatant water (dissolved Cu and Zn 0.005-0.300 and 0.001-0.100 mg/L, respectively). Tests showed that all fish survived the 96H exposure period, indicating that there was no acute toxicity associated with the mine's tailings-pond water (Rescan 1992a). Most frequent tests were conducted since 1985 when intermittent discharges of supernatant water to Rum Bay (Hagan Arm) were authorized pursuant to the B.C. provincial Waste Discharge Permit (#1505) for the Bell mine. As noted previously, a cursory review of Environmental Monitoring reports showed that all bioassays (over 20 in total) showed the effluent was non-toxic at the time of discharge.

Microtox (20 and 30 minute EC_{50}), rainbow trout (96H LC_{50}) and *Daphnia magna* (48H LC_{50}) tests were conducted at 4 sites on a bi-weekly basis in a study to assess the use of Microtox as an alternative assay in the acute toxicity assessment of (Bell) mine effluent (Mochon and Prairie 1992). For 20 minute Microtox tests the EC_{50} average dissolved Cu concentration was 0.31 m/L compared to 0.55 mg/L rainbow trout (LC_{50}) and 0.17 mg/L for *D. magna* (48H LC_{50}). The results indicate merit in pursuing the use of Microtox as a possible means for routine toxicity screening of mine effluents.

Sediment bioassays were conducted by Environment Canada in 1990 using the Chironomus tentans emergence tests (Godin *et al.* 1992). It was found that the Rum Bay and Hagan Arm samples showed delayed fly-emergence effects. The authors could not confirm the cause of these effects since metals (i.e., exchangeable Cu, Cd, Ag, Pb, Zn from sequential extraction) were below detection limits. A sediment sample from Granisle Bay (near McDonald Island) showed acute toxicity where high concentrations of exchangeable aluminum (450μg/g), copper (111 μg/g) and zinc (90.3 μg/g) from sequential extraction of sediments were recorded (Godin *et al.* 1992).

d) Benthos

From a review of historical data (Rescan 1992a), benthic invertebrate communities in some near-shore areas around Newman Peninsula exhibit reduced abundance and diversity in species composition relative to other sites. Data collected in 1992 (Rescan 1992b) indicated increases in the composition of facultative and sensitive taxa for the Main Arm of Babine

Lake compared to historical baseline data, and there were marginal shifts from tolerant to facultative fauna for Hagan Arm.

Diversity indices for two shallow sites in Hagan Arm were indicative of healthy benthic populations, and indices for samples obtained from sites in Woolverton Bay were moderately good. Commonly, however, the sampling sites were not consistent from study to study (Rescan 1992a). In light of the variation between sites and the limited amount of data available, it is difficult to determine whether the differences reflect natural site-to-site variability or the presence of contaminated sediments.

The results of the Godin *et al.* (1992) benthic invertebrate study did not compare fully with those of earler studies (e.g., Hatfield 1989, as reported in Rescan 1992b). Diversity indices were lower, eveness was more consistent, and richness was higher than reported by Hatfield (1989). There were, however, some important differences between the studies. Godin *et al.* conducted their studies at a different time of year (September, versus June), collected fewer samples (single grab, versus triplicates) and had slightly different sampling sites than Hatfield. The authors also indicated that mesh size could have had an effect on the results.

e) Phytoplankton and Zooplankton

A review of historical data indicates that there is considerable seasonal, spatial, and temporal variability in phytoplankton biomass (as reflected by chlorophyll-a levels) in Babine Lake. Mean chlorophyll-a levels were similar in both Hagan Arm and the main body of Babine Lake. A comparison of 1991 sampling results and historical data (Rescan 1992b) suggested no apparent effect from past mining activities. Total densities and densities of individual zooplankton species were variable from site to site in Babine Lake, but the differences were small.

The 1991 analyses of zooplankton tissue for trace metals confirmed previous analyses which showed that, for selected metals (including Cu, Pb, and Zn), concentrations in zooplankton tissues from sites in Hagan Arm near the mine workings and discharge outfall were higher than those from sites in the Main Arm. The similarity of zooplankton density and species composition in Hagan Arm and the main body of Babine Lake suggests that there have been no obvious negative impacts on zooplankton as a consequence of the mining activity. However, as replicate and statistical comparisons were not conducted and seasonal variability was not controlled between years, temporal trends could not be assessed.

f) Fish

Metal levels in muscle tissue of fish from Babine Lake are generally within the range reported for fish in unpolluted waters. In general, there were also no substantial differences in fish-tissue metal levels in Hagan Arm compared to the Main Arm (Rescan 1992 (a) and (b)). However, with the exception of burbot, concentrations of Cd, Cu, and Zn in liver tissues of fish from both arms are higher (200-300 ppm) than those reported for livers of fish from unpolluted waters (1-30 ppm) (Rescan 1992b; Godin *et al.* 1985).

Analysis of Molecular Indices of Metal Stress (MIMS) of liver tissues from one lake trout and one burbot, conducted in 1991-1992, suggested little evidence of stress from Cu uptake in these species. Metallothionein concentrations from several lake trout and burbot livers from Babine Lake also suggested that the specimens were healthy and not adversely affected by the mining operation. It was further suggested that any effects noted could be attributed to environmental or physiological stresses rather than ambient Cu concentrations which were believed to be reduced by the high Cu-complexing capacity of Babine Lake waters (Rescan 1992b). However only a limited number of samples were tested, and there are few referenced results for these species of fish with which to make such broad conclusions. Nevertheless, test methods such as MIMMS and metallothionein measurements may provide useful information about sublethal effects of effluent discharges and should be considered for inclusion in monitoring programs, particularly in situations in which no obvious acute lethality effects of effluent discharges are evident..

Numbers of several economically important fish species (e.g., rainbow trout, lake trout, kokanee) have been on the decline in recent years. There is no clear understanding of the cause of this phenomenon, but it has been suggested that the success of commercial sockeye stock improvement could have placed pressure on food (e.g., zooplankton) availability (Rescan 1992a).

Although some limited sampling of fish was conducted in the vicinity of the Bell minesite, the primary purpose of the studies was not to define the overall status of fish populations but to collect specimens for physiological assessment and/or analysis of metals in tissues. Accordingly, the data available are insufficient for evaluation of the potential impacts of the mine on fish populations.

2.5 Discussion

2.5.1 Causes and Effects Analysis

Whereas it is debatable whether the mining of the Bell Copper deposit has led to any long-term negative impact on the health of the Babine Lake ecosystem, there is strong evidence that the observed changes in abiotic parameters (i.e., water and sediment chemistry) are related to mining activity. Admittedly, natural weathering of mineralized rocks near the shoreline of Newman Peninsula may have contributed to the elevated Cu content of some shallow sediments. Nevertheless, the proximity and areal extent of identified anomalous sediments to mine workings, infrastructure facilities, and the effluent outfall strongly suggest a predominantly mining-related cause. Similarly, the short-term differences in water chemistry, especially in dissolved-Cu concentration, between Hagan Arm and the main body of Babine Lake observed since the early 1980s is most likely caused by mining activity.

The Hagan Arm basin is separated from the main Babine Lake basin by a shallow shelf near McDonald Island which hinders rapid mixing. Within Hagan Arm itself, water in Rum Bay has shown higher ionic strength than the main body of Hagan Arm (Rescan 1992a), reflective of the influence of tailings-supernatant discharge. The observation that the measured Cu concentrations in Rum Bay are not statistically different from the rest of Hagan Arm can probably be explained by the relatively low concentrations and higher diffusivity of the metal ion than for anions such as sulfate and chloride. The apparent recent improvement in water quality in Hagan Arm (Figure 2.2), concomitant with continual upgrading of seepage-collection systems at Bell mine, further substantiates the connection between water chemistry in Hagan Arm and mine-effluent control at the mine operation.

In contrast, the biotic effects of mining at Bell are somewhat more difficult to ascertain. The elevated metal content in tissues of zooplankton collected from sites in Hagan Arm near mine workings suggests possible mining-related effects. The generally high Cu content in liver tissues of most fish species in both Hagan Arm and the main part of Babine Lake (relative to other areas), however, suggests that prevalent anomalous concentrations in background Cu in the lake may be the cause of elevated metal levels in fish tissues. Available toxicity data for the tailings pond supernatant indicate that acute toxicity of the effluent discharge is not a factor. Some seepage collection pond waters are extremely high in copper (Noranda Minerals, 1985 to 1993) and pit water has on occasion been toxic to fish (Environment Canada 1982) however, these waters are not purposely discharged to the environment.

Benthic invertebrate communities in some near-shore areas around Newman Peninsula are characterized by reduced abundances and species diversity compared with other sites (Rescan 1992a), but there are insufficient data to differentiate site variation from contaminant effects.

Other biological information (e.g., zooplankton, phytoplankton) suffers similarly from the absence of studies of long-term trends specific to the Bell minesite.

Although some limited fish-capture data have been collected in the vicinity of the Bell minesite, the data are of marginal value in describing the overall status of fish populations. The primary purpose of such sampling commonly was to obtain specimens for analyses, e.g., body metal-burdens rather than to determine populations per se. Given the size of Babine Lake, the relatively good quality and limited quantity of effluent discharged, the other factors which can affect fish populations in the lake (e.g., commercial and sport-fishing pressures, salmon enhancement, ocean survivability), and the limited data specific to the Bell mine, it is considered unlikely that the effects from the mine operation could be defined with any confidence at this time.

2.5.2 Adequacy of Monitoring Program

The documents reviewed have been prepared by different agencies and were designed to address specific issues (eg. temporary supernatant discharges) and/or different stages of the mining development. Accordingly, the studies commonly used different methods, sampling frequency and sites. The resultant data-set therefore lacks the consistency of single purpose and comprehensive environmental monitoring program.

2.5.3 Applicability of MMLER and Provincial Regulatory System

The Bell Copper mine began operations in 1972, before promulgation of the MMLER. Accordingly, the mine was subject to the Guidelines when they came into effect in 1977. Furthermore, prior to the temporary closure in 1992, the Bell mine utilized a closed-circuit processing system, with no direct effluent discharge into the receiving environment. Thus, the effectiveness of the MMLER/G could not be determined.

The need to discharge tailings-pond supernatant during the temporary closure in 1982-85 and the subsequent discharge into Hagan Arm on an ongoing but intermittent basis since 1985 were governed pursuant to a B. C. provincial Waste Management Branch permit. The conditions specified in the permit were more stringent (0.05 mg/L dissolved Cu) than the MMLER (0.6 mg/L total Cu). Again, the effectiveness of MMLER in protecting the aquatic resources at Babine Lake cannot be tested for the case of Bell Copper mine.

Natural background concentrations of Cu in Babine Lake frequently exceed established criteria for protection of aquatic life (Figure 2.2). Studies have shown that the Cu-complexing capacity of water in Babine Lake exceeds current background concentrations

of the metal (Rescan 1992 (b)) and this may assist in reducing environmental stresses imposed on resident biota. However, long term benefits of metal complexation on the more sensitive organisms are necessary to confirm the hypothesis.

The MMLER are based on Best Practical or Best Available technologies, with final effluents regulated on concentrations of a limited number of parameters. Furthermore, the Regulations do not require an effluent to be non-toxic, although provisions are made for toxicity testing under the Guidelines and the general provisions of the *Fisheries Act*. There also appears to be some uncertainty about whether the MMLER are applicable to closed or abandoned operations. Given that the Bell mine did not discharge effluent directly to the receiving environment during the operational phase of the mine, and when discharges were necessary they were always non-acutely lethal (i.e. 96h LC_{50} > 100%) it was therefore difficult to evaluate the effectiveness of the current MMLER to the Bell operation.

Although providing a basic level of effluent treatment and control the MMLER do not therefore necessarily provide adequate protection for the aquatic environment in all cases. Certainly, as noted elsewhere in the case study, there is no evidence to indicate that the Bell mine had a significant impact on Babine Lake. However, it may be necessary to consider effects of metal combinations, effluent loading and employ more sophisticated assessment techniques to better define effects of the mine. This is currently not required by the MMLER.

The B.C. provicincial Mining Objectives (B.C. MOE 1979) are similarly based on treatment technology, but the waste-management permitting system provides greater flexibility to consider site-specific environmental conditions (e.g., complexation capacity), and a broader range of regulatory requirements (e.g. contaminant loading as well as concentrations) and effluent and receiving environment monitoring.

2.6 Conclusions and Recommendations

Copper is the principal metal of concern associated with the operation of the Bell Copper mine. As the British Columbia provincial regulations are more stringent than MMLER, the effectiveness of the latter in protecting the aquatic resources in Babine Lake cannot be tested in this case study. A critical review of environmental data collected in the vicinity of the Bell Copper mine suggests that the mining development has led to a change, albeit possibly temporary, in the water chemistry of Hagan Arm (especially for dissolved Cu) compared to that of the main part of Babine Lake. Elevated Cu concentrations in the top sediments at several sites adjacent to the mine workings have also been documented. The proximity of the Granisle mine may have a confounding effect on the interpretation of the study results for the Bell mine, especially for those sites closer to Granisle.

The high Cu-complexing capacity of the Babine Lake water may help to reduce potential stresses on resident biota. As most discharges from the mine have been non-acutely toxic, no acute toxicity effects would be expected in the receiving environment. However, extensive studies of sublethal and chronic effects have not been conducted.

Environmental effects monitoring programs should include adequate baseline monitoring of background metal levels and complexing capacity. Given the importance of such data in defining behaviour of contaminants (e.g. metals) in the environment they should be considered in evaluating the risks posed by proposed mine effluent discharges and in developing site specific mine regulatory requirements. Furthermore, more research is also required to defining chronic and sublethal toxicity effects of metal contaminants, and on the application of ecosystem approaches (eg. mesocosm studies) to routinely assess sublethal and chronic effects of mining effluents.

As a result of the history of the Bell Coppper mine operation (i.e., no positive discharge during most of the operating life of the mine), there was limited pre-mine receiving-water information. Furthermore, much of the available information was collected for different reasons (e.g., spill monitoring, temporary discharges) using a variety of techniques. Commonly lacking is sufficient frequency of sampling, and coordination of the monitoring sites, e.g., for water quality, sediment, benthos, fish. There is a need to include both temporal and spatial reference sites in the design of monitoring programs.

2.7 References

*Basu, K. (1984) Literature review and data summary of studies related to Bell mine, Babine Lake. Environment Canada Pacific and Yukon Region, Regional Program Report 84-10, 81 p. (AQUAMIN Report 124).

B.C. MOE (1979). Pollution Control Objectives for the mining, mine-milling and smelting industries of British columbia. B.C. Ministry of Environment, Victoria, B.C.

B.C. MOE (1982). Letter of transmittal from B.C. Ministry of Environment to Maclaren Forest Products Inc. re: amended Permit No. PE-1505. October 31, 1982.

B.C. MOE (1983) . Letter of Approval SK-20-AE. October, 1983.

B.C. MOE (1984). Letter of transmittal from B.C. Ministry of Environment to Maclaren Forest Products Inc. re: amended Permit No. PE-1505. November 20, 1984.

B.C. MOE (1989) Correspondence from B.C. Ministry of Environment to Noranda Minerals Inc. Bell mine. December 18, 1989.

Carson, D.J.T., Jambor, J.L., Ogryzlo, P.L. and Richards, T.A. (1976): Bell Copper: geology, geochemistry and genesis of a supergene-enriched biotitized porphyry copper deposit with a superimposed phyllic zone. CIM Special Volume 15, 245-263.

Environment Canada (1982) Letter from B.A. Heskin (DOE) to R.H. Ferguson (B.C MOE) re: proposed amendments to Permit PE-1505. Sept. 28, 1982.

Environment Canada (1990) Letter from M. Ito to M. Lengwenus (Noranda Minerals Inc.). June 19, 1990.

Godin, B., Hagen, M. and Mitchell, G. (1992) Babine Lake monitoring June 19-22, 1990. Environment Canada Pacific and Yukon Regional Data Report DR 92-10, 35p.

*GODIN, B., ROSS, M. and JONES, M. (1985) Babine Lake: data from chemical and biological surveys May '83, May '84, July '84. Environment Canada Regional Program Report 85-14, 167 p. (AQUAMIN Report 545).

Kennedy, P. (1983a) The Month of May Report on the Discharge of Supernatant from the Tailings Impoundment of MacLaren Forest Products Inc., Babine Mining Division, Bell Mine into Hagan Arm of Babine Lake Under Letter of Approval No. SK-20-AE.

Kennedy, P. (1983b). The Month of June Report on the Discharge of Supernatant from the Tailings Impoundment of MacLaren Forest Products Inc., Babine Mining Division, Mine into Hagan Arm of Babine Lake Under Letter of Approval No. SK-20-AE.

Kussat, R.H., Knapp, W.D. and McIndoe, R.A. (1972), Progress Report: Evaluation of the Environmental Impact of Mining, Community Development and Logging Activities on the Water Quality of Babine Lake. Dept. of Environment, Fisheries Serv., 34p.

McArthur, D.R. (1995). Project Manager, Bell Copper Mine. Personal Communication.

* Mochon, A., R. Prairie (1992). A Study on the Use of Microtox as an Alternative Assay in the Acute Toxicity Assessment of Mine Effluent. Report prepared for Noranda Minerals Inc., 34 p. plus figures.

Noranda Minerals Inc. (1986). Bell Mine-Environmental Report, 1985. Submitted Pursuant to Permit No. PE-1505. March 1986. 7 p. + Addenda.

Noranda Minerals Inc. (1987). Bell Mine-Enviro[illegible]ntal Report, 1986. Submitted Pursuant to Permit No. PE-1505. May 1987. 8 p.

Noranda Minerals Inc. (1990). Bell Mine-Environmental Report, 1989. Submitted Pursuant to Permit No. PE-1505 35 p. + Figures.

Noranda Minerals Inc. (1991). Environmental Report, 1990. Submitted Pursuant to Permit No. PE-1505. 14 p. + Appendices.

Noranda Minerals Inc. (1992). Bell Mine- Environmental Report, 1991. Submitted Pursuant to Permit No. PE-1505. 35p. + Figures and Appendices.

Noranda Minerals Inc. (1992b). Company brochure, "Bell Mine Environmental Program", provided for field trip associated with the 16th Annual- British Columbia Mine Reclamation Symposium, Smithers, B.C. June 1992.

Noranda Minerals Inc. (1993). Bell Mine-Environmental Report, 1992. Submitted Pursuant to Permit No. PE-1505. Dec. 1993. 34 p. + Tables and Appendices.

Noranda Minerals Inc. (1994). Bell Mine-Environmental Report, 1993. Submitted Pursuant to Permit No. PE-1505. June 1994. 33 p.

*Rescan Environmental Services (1992) Bell 92 Project Closure Plan Support Document G Historical Environmental Data of Babine Lake. Unpublished Report, 24 p. plus appendices (AQUAMIN Report 273).

*Rescan Environmental Services (1992) Bell 92 Project Closure Plan Support Document H Existing Environmental Conditions of Babine Lake in the Vicinity of Bell mine. Unpublished Report, 36 p. plus appendices (AQUAMIN Report 274).

***AQUAMIN Reports**

24: Basu (1984)
273: Rescan (1992a)
274: Rescan (1992b)
45: Godin *et al.* (1985)
4: Mochon and Prairie (1992)

3 Brenda

3.1 Introduction

The Brenda copper-molybdenum mine, which produced from 1970 to 1990, is at lat. 49° 52' 30"N, long. 120° 00'W, in the Okanagan watershed approximately 22 km northwest of Peachland and south of Kelowna, in the Okanagan and Nicola valleys. The minesite is situated at 1460 m altitude, and occupies a divide between the headwaters of Trepanier Creek and Peachland Creek, both tributaries to Okanagan Lake (Figure 3.1). The surrounding area is characterized by gently rolling, forested upland, with scattered glacially rounded outcrops. The site receives approximately 640 mm of precipitation annually, two-thirds of which falls as snow. Summers are short, with a frost-free period of about 60 days, and winters are long and cold.

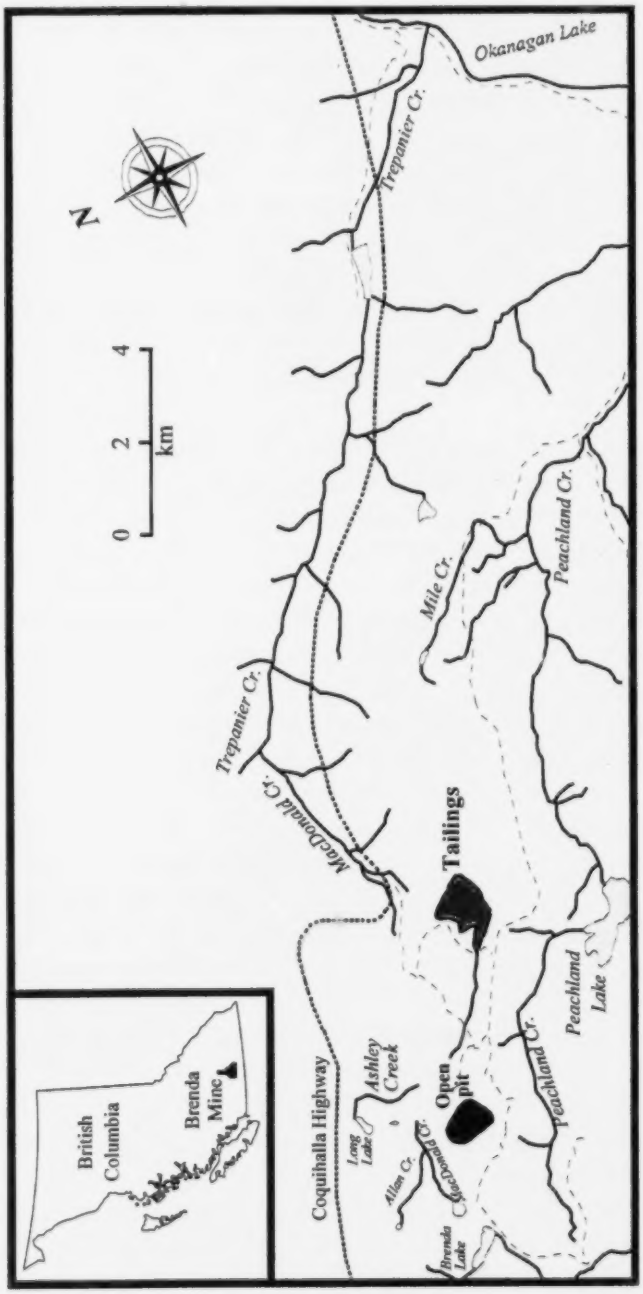
The Brenda deposit occurs within a composite, zoned quartz diorite to granodiorite stock of Jurassic age that intruded Upper Triassic sedimentary and volcanic rocks of the Nicola Group (Soregaroli and Whitford 1976). Pre-ore and post-ore dykes with widely divergent compositions cut the stock. Mineralization is largely confined to veins, and disseminated sulphides are sparse. At least five stages of veining, each with unique directions and mineralogy, were developed in fractures created by east-west regional compression. Grade of mineralization is a function of fracture density and mineralogy of the veins. Chalcopyrite and molybdenite, the most abundant sulphides, are accompanied by minor pyrite and magnetite hosted by quartz, K-feldspar, biotite, epidote, and calcite.

a) History

Tune-up of the flotation concentrator commenced in November 1969, and the mine operated until 1990. The mine produced 278,000 tonnes of Cu and 66,000 tonnes of Mo in concentrate form, and employed between 350 and 400 people during the operation.

The primary features of the Brenda minesite include a large open pit from which ore and granodiorite waste rock were excavated, four rock stockpiles which total 109 Mt and are at the periphery of the pit, a concentrator building and other surface facilities, and a tailings impoundment containing 150 Mt of finely ground flotation tailings permanently stored behind a dam constructed of 50 Mt of coarse tailings sands; there are also numerous pipes and ditches used for routing surface runoff in the vicinity of the site, and several roads.

Figure 3.1: Location of the Brenda Mine



The mine operated under a zero-discharge permit. This was achieved through the diversion of practically all surface water flowing towards the mine area. Water directly affected by mine operations was discharged to the tailings impoundment, and recycled to the mill for reuse. The recycle rate was 90%, and seepage from the tailings embankment and saddle dam was pumped back to the tailings pond.

Runoff from the granodiorite rockpiles was initially released to the local environment. Because of the alkaline nature of the waste rock, dissolution of low concentrations of Mo in the runoff waters occurred and became evident by the late 1970s. During operations, the runoff oversupply was contained by the rate of tailings-dam building. In 1984, because of a shutdown of the mine for economic reasons, a limited discharge from the site was required to protect the integrity of the tailings dam.

b) Milling Methods

Processing of the ore involved the use of a copper-molybdenum flotation circuit, followed by a molybdenite leaching process using ferric iron. Because the ferric iron leaching process was used, Brenda process water generally had a lower Mo content than that of the rockpile water. The spent leach brine was discharged to the tailings pond, at which the iron precipitated as the hydroxide. Some of the Mo was coprecipitated with the iron, thus resulting in the lowered Mo concentration.

The 109 Mt of granodiorite waste rock stored at the periphery of the pit contains approximately 0.10% Cu as chalcopyrite, and 0.015% Mo as molybdenite. The approximately 200 Mt of tailings average 0.015% Cu and 0.005% Mo.

c) Effluent

Water affected by the mine operations was discharged to the tailings pond and then recycled to the mill for reuse in the flotation process. Excess water accumulated in the tailings pond and was contained within the pores of the tailings solids stored in the tailings impoundment. Following closure of the mining and milling operations in 1990, 80% of the 15 M m^3 of process water stored in the tailings pond was pumped to the open pit for storage.

During the operation, tailings from the concentrator were cycloned; the sands were used for dam building and the slimes were used to produce an impervious beach area. During the operation 10 M m^3 of water per year were recycled from the mill to the tailings pond. Since the mine closure, recycling of water has ceased, but the tailings pond has been used as collection area for minesite runoff. Approximately 1.5 M m^3 per year of this water is pumped

annually from the tailings pond to the open pit.

d) Effluent Treatment

By 1979, a series of diversion and collection ditches had been constructed to direct all runoff water from the rockpiles to the tailings impoundment, resulting in surplus volumes of on-site surface waters. Nitrate-contaminated water was piped into the tailings pond. Waters with high nitrate values also contained high levels of dissolved Mo. After completion of the diversion system in 1980, nitrate concentrations decreased. Following an evaluation of water-treatment technology methods, remediation work between 1993 and 1994 was concentrated on the biological or wetland process, and on the iron-precipitation process, to meet the drinking-water objective of 0.25 ppm Mo.

e) Seepages/Runoff

Runoff from the rockpiles was initially released to the local environment, but because of the alkaline nature of the waste rock, dissolution of low concentrations of Mo in the runoff waters occurred, and became evident by the late 1970s. Water in this area will flow in perpetuity because of annual replenishment by precipitation Contamination is attributed to the water flowing through the waste rock and dissolving the various minerals. The volume averages 0.7 to 1 M m^3 of water annually, with the major flows occurring during the spring runoff.

f) Compliance Record

Although data on acid mine drainage are not cited in the reports reviewed, high nitrate values (5 - 30mg/L) were recorded from 1975 to 1978 at Site 353 at MacDonald Creek (Bryan 1991). These concentrations stimulated algae growth downstream. As well, the monitoring results (Bryan 1991) for Trepanier Creek showed a slight influence of tailings leachate from the mine. Concentrations of Mo were higher downstream than upstream of Macdonald Creek, and mean values of sulphate and filterable residues had been affected by input from Macdonald Creek.

The MMLER came into effect in 1977, at which time the Brenda operation was defined as an "existing mine", to which only the Effluent Guidelines applied. Although these guidelines were met in the Mine effluents, the most critical contaminants, Mo and nitrates, aren't regulated by the MMLER.

g) Additional Information

- the Mo content of Peachland Lake, which is part of the Peachland Watershed and is the water reservoir for the Municipality of Peachland, approached the Canadian Drinking Water Standard of 0.25 ppm

- because of the limited amount of industrial land available in the rapidly growing Okanagan Valley, and because of the proximity of the site (via the Coquihalla Connector Highway) to Vancouver, a modified land-use proposal for an industrial park has been identified

- levels of Cu and Mo in the grasses and legumes are monitored annually. Plants grown in these soils with elevated metal levels can accumulate them in their tissues. This has not resulted in toxicity in plant growth, but may cause toxicity in ungulates if the animals are fed these plants as a constant diet

- a disease known as molybdenosis can develop in cattle, particularly under conditions of inorganic sulphate and Cu deficiencies

- problems arise in determining water-quality objectives for the proposed minesite effluent. Objectives for aquatic life would limit Mo content to <1 ppm, and drinking-water-quality objectives lower this to 0.25 ppm. The irrigation objective, set for poorly draining land used to grow forage crops, proposes a limit of 0.01 ppm

- scientific literature indicates that forages containing 10 to 20 mg/kg Mo dry weight can be toxic to cattle and sheep. The literature further indicates that cattle have been adversely affected when grazing on pastures in which Cu:Mo = <3:1, or when fed low-Cu diets containing 2 to 10 mg/kg Mo; this issue is currently under study at the Valley Copper (Highland Valley) and Brenda mines

- agronomic plants grown on the tailings have low ratios of Cu:Mo, the mean Cu_{total} levels are in the range of 6 to 12 mg/kg, and the mean Mo_{total} levels range from 55 to 191 mg/kg. The mean Cu_{total} levels in plant foliage collected from the rock piles range from 14 to 17 mg/kg, and the mean Mo_{total} levels range from 114 to 643 mg/kg.

h) Adequacy of Monitoring Programs

The adequacy of the study/experimental design, reference/controls, study methods, quality control, and data analysis are evaluated as part of the Results Assessment.

3.2 Results of Aquatic Effects Assessment

a) Water Quality

An extensive water-quality sampling program of the lakes and watercourses in the area of influence of Brenda Mines was undertaken for a twelve-month period from the autumn of 1990 to summer 1991 (Rescan 1993a). The program included Peachland, MacDonald, and Okanagan lakes, and the Peachland and Trepanier Creek drainages. Sampling was conducted at sites both above and below mine influence to provide a baseline for developing the environmental impact assessment of the decommissioning plan (providing water-quality data as a function of sampling-station location). As well, Bryan (1991) described some effects of Brenda Mines wastewater seepage from waste rock and the tailings pond; this water-sampling program included the MacDonald, Trepanier, and Peachland Creek drainages and Peachland Lake.

Water-quality criteria established by the British Columbia Ministry of Environment, Lands and Parks (MELP) were used to assess the significance of the water-quality data and to establish site-specific water-quality objectives. The criteria, modified to fit locally important water uses and to use ambient water quality, were used to develop MELP Provisional Water Quality Objectives for west-shore tributaries to Okanagan Lake, including the Trepanier and Peachland drainages.

The only assessment of data quality conducted for the 1990/91 monitoring program (Rescan 1993a) was the collection of duplicate samples for inter-laboratory comparison. Although it was indicated that the "analyses from both labs compared favorably", no data or criterion to evaluate acceptability were provided. No assessment of sampling precision or blank quality, or analytical precision and accuracy, was provided. Therefore, the accuracy and reproducibility of these data may not be adequate for comparison between sites, or for comparison with the Water Quality Objectives. Similarly, as no quality-assurance data were reported for the values presented in Bryan (1991), a similar caveat applies to the interpretation of those water-quality data.

Peachland Lake has a surface area of approximately 109 ha. Its outlet is dammed, and releases of water to the lower reaches of Peachland Creek are controlled. The water quality of Peachland Lake, with respect to Mo levels, was reported to have improved following the construction of the water-collection system and diversion works that routed minesite runoff to the tailings pond. The lake water is relatively soft, and mildly alkaline. Concentrations of dissolved metals were generally low; the exception was that levels of Mo were sufficiently elevated (at approximately 0.03 mg/L) relative to waters of the surrounding drainage basin (at approximately 0.004 mg/L), so that the differences are likely to be real rather than artifacts of poor data quality. Bryan (1991) reported Mo levels that exceeded the provincial

water-quality criteria for protecting drinking water (0.25 mg/L) and irrigation water (0.01 mg/L), but the levels did not stress fish in short-term laboratory exposures.

MacDonald Lake is a small, shallow lake situated northwest of the Brenda mine and upstream of it. Mo levels at the 0.001 mg/L detection limit were recorded.

Okanagan Lake, covering an area of 360 km^2, the eighth largest freshwater body in British Columbia, is the predominant feature of the Okanagan Basin. The lake is large, deep, and monomictic, with a well-developed thermocline that persists throughout the summer and fall. Okanagan Lake has been sampled twice annually, at which time the Mo levels were generally detectable but consistently below the provincial criteria.

The water quality of the MacDonald Creek settling pond was influenced by its proximity to the tailings dam and lower reclaim pond in that water seeping from these structures may enter the pond. The water of this section is hard, and is high in dissolved solids. MacDonald Creek discharges into Trepanier Creek, and makes up 2.5% of the flow in Trepanier Creek. However, the physical/chemical conditions of MacDonald Creek were similar to those occurring in the two larger lakes. Noticeable exceptions included seasonally elevated conductivity, turbidity, and suspended-solids levels, and concentrations of Mo about equal to the provincial criterion for irrigation use.

Bryan (1991) reported that Site 353, located downstream of the open pit on the MacDonald Creek diversion, became contaminated by minerals and blasting residues as the waste-rock pile expanded. Downstream, however, these concentrations diminished to below detection limits. At the reclaim dam, which is upstream of all tailings, Mo and filterable residue were high because of runoff and leachate from the waste rock. Except for Mo, Cu and Zn, other dissolved heavy metals were usually below detection limits.

Reported water quality in Trepanier and Peachland creeks was generally good. Hardness increased in both creeks in a downstream direction, reaching nearly 100 mg/L (as $CaCO_3$) at Okanagan Lake. Several data sources indicate that natural background concentrations of Cu exceeded the provincial criterion for the protection of aquatic life in both creek systems. Molybdenum levels in Peachland Creek below Peachland Lake reflected the elevated concentrations in the lake, which at 0.045 mg/L to 0.103 mg/L were significantly above background levels (about 0.01 mg/L). Since 1990, however, Mo in Peachland Lake has generally fallen to below 0.03 mg/L (D. Perkin, pers. omm. 1995). Zinc levels were reported as much higher in the creeks (0.011 mg/L to 0.08 mg/L; Bryan 1991) than in the lakes (<0.01 mg/L; Rescan 1993a). Although this discrepancy may be due to the inclusion of 1972 to 1990 values in Bryan (1991), and only 1990 and 1991 values in Rescan (1993a), differences in data quality cannot be ruled out without the supporting information.

Bryan (1991) indicated that the results of monitoring showed a slight influence of tailings leachate from the mine. However, the slight increase in Mo levels from the downstream location on Trepanier Creek to the upstream upstream site on MacDonald Creek cannot be adequately assessed without information on the data quality.

b) Sediment Quality

Duplicate stream-sediment sampling from selected sites in Trepanier, MacDonald, and Peachland creeks, as well as surface grab sampling from MacDonald, Peachland, and Okanagan lakes, was reported by Rescan (1993a). Concentrations of Cu and Mo in the bottom sediments of Peachland Lake were compared among various stratigraphic levels in a core profile (Rescan 1992a).

No data on quality assurance were reported for the sediment samples collected by Rescan (1993a). Laboratory precision and accuracy were both assessed on sediment analyses reported in Rescan (1992a), but the results for precision were not included in the report. Accuracy of the X-ray fluorescence analysis of Mo, Cu, and other metals in solids was assessed by comparison of single measurements with known single reference values (means?); however, no criteria for determining acceptability were included. The analytical accuracy of porewater Cu was acceptable, based on a single measurement of a certified reference water sample. The accuracy of porewater Mo values was not assessed. No data on precision were provided.

Although the lake and stream-sediment metal values reported in Rescan (1993a) were obtained from analysis of the minus 80-mesh (~200 μm) fraction, the sampling technique was capable of detecting only gross enrichment from mine wastewater releases. Assuming that the lake-bottom samples were taken from similar depositional environments (based on the water depths listed in Table 3-4, page 3-16), use of a Ponar sampler will not enable inter-lake comparisons of metal enrichment to be made.

Although significant differences are evident in Mo, Cd, and Cu between Peachland Lake and Okanagan Lake, differences in total organic carbon (% loss on ignition) were also recorded, indicating different metal-adsorption capacities between these water bodies. As lakes with sediment of such variable organic content may also have different sediment-accumulation rates, Ponar samples may contain variable mixtures of metal-enriched sediment and sediment belonging to the less contaminated, pre-development stage. To draw conclusions on the extent and timing of metal enrichment, and thus relate it to mine-waste releases, requires the establishment of profiles of the metals in the sediment core.

Analysis of sediment cores from the centre of Peachland Lake in 1992 (Rescan 1992a)

showed Mo enrichment of about 10× relative to deeper strata in the core. It was postulated that this high degree of enrichment, which occurred in the strata at a depth of 4 to 8 cm, was attributable to the discharge of contaminated minesite runoff during the 1970s. Although this seems to be a reasonable inference in light of the return to pre-enriched levels in the top 4 cm of the sediment profile, no dating was done to confirm the chronology. Relatively greater Mo levels in sediment porewater taken from the top stratum of the core may mean that the sediment will not act as an effective sink for future increased Mo releases from site drainage. Thus, future releases of greater Mo loadings may cause enrichment of downstream water and sediment.

c) Benthic Macroinvertebrates

Conclusions on the effects of the mine wastewater and runoff releases could not be drawn because of the high variability in the benthic invertebrate database (Rescan 1993b). Replication and stratification of sampling sites, and the use of controls were inadequate to make quantitative comparisons. The same problems occurred in the study by Bryan (1991), in which there was a lack of replication and temporal controls. The data in these studies are useful only as pilot information for the design of an adequate baseline or monitoring program. No work on lake samples was done even though, given the evidence for enriched Mo and Cu in sediment, these might be expected to show measurable effects on benthic invertebrates. Given the evidence of potential enrichment of metal levels in sediment in Peachland Creek below Peachland Lake (Rescan 1993a), it would be worthwhile establishing an adequate sediment and benthic invertebrate baseline in this stream section.

d) Fish

Variations in fish-community compositions and populations are confounded by harvesting and habitat-disturbance factors (Rescan 1993b). Stream channelization and flow reductions in Trepanier and Peachland creeks confound mine effects as causes of differences in rainbow trout populations (Rescan 1992b).

e) Periphyton

High habitat variability (stream flow, water depth, and substrate composition) confounds the reported differences in periphyton densities and community composition between sites (Rescan 1993b). Density reductions of about 60× at downstream sites on Trepanier Creek, relative to upstream controls, indicate that a better designed, more sensitive periphyton baseline has potential for the monitoring of future effects.

3.3 Evidence for Aquatic Effects

a) Effects on Use of Water and on the Aquatic Environment

The Brenda mine operations changed water quality in the streams and in Peachland Lake between 1970 and 1980, during which the release of waste-rock runoff water was allowed. The most notable effect occurred during the 1970s, when Mo concentrations in Peachland Lake exceeded guidelines for potable drinking water. The problem was recognized by the mine, and lake-water quality improved with respect to Mo levels following the construction of the water-collection system and diversions that routed site runoff to the tailings pond. Frequently, however, the Mo levels in water near the mine are above the guideline for irrigation water to be used in forage fed to cattle.

Levels of Cu and Mo in the grasses and legumes are monitored annually. Plants grown in these soils with elevated metal levels can accumulate the metals in the plant tissues. This has not resulted in toxicity in plant growth, but may cause toxicity in ungulates if the animals are fed these plants as a constant diet. A disease known as molybdenosis can develop in cattle, particularly under conditions of inorganic sulphate and Cu deficiencies.

The undiluted mine wastewater was occassionally toxic to invertebrates, but not acutely toxic to fish. Through monitoring and development of the water-collection system and diversion works, most of the mine wastes were kept out of flowing waters, mitigating the potentially harmful effects on the organisms living in the streams or lakes around the mine.

b) Effects on the Benthic Environment

Between the time of dam construction and 1979, runoff into Peachland Lake from the nearby Brenda minesite likely accounted for a pronounced enrichment of Mo in the sediment of the lake. Solid-phase Mo levels are an order of magnitude lower in the top 3cm of the deposits, indicating that the diversion of the runoff since 1979-80 was likely responsible for returning Mo concentrations in sediment to background levels. Copper was the only other metal enriched in the cores of Peachland Lake sediment (Rescan 1992a), with levels only about twice the inferred background.

Monitoring of sediment chemistry in the other lakes and creeks is less definitive of effects because the results from sampling only the surface sediment cannot be compared with results for other locations with greatly varying sediment types, including variations in organic carbon content (Rescan 1993a; Table 3-4, pages 3-16). The sediment-core data available from Peachland Lake indicate that significant Mo enrichment was buried in core strata deeper than that which may have been penetrated by the Ponar sampler. The Mo values in

surface-sediment samples taken in 1993 (3 μg/g to 79 μg/g) are within the background range for Peachland Lake sediments.

None of the available benthic invertebrate data provided clear indications of effects from the mine wastes. Although the designs of the two benthic invertebrate programs, and particularly the lack of adequate replication and statistical design, would likely allow detection of only gross effects, the lack of effects may also be attributed to the lack of significant sources of mine contamination at the time of the studies. However, the evidence for significant Mo contamination of water and sediment from surface runoff suggests that there is a potential for future effects unless minesite drainage is adequately controlled.

3.4 Assessment of the Effectiveness of the MMLER and Provincial Regulatory System

Although the surface runoff that was released in the 1980s from Brenda mine is technically "effluent" under the MMLER, only the Effluent Guidelines applied because the operation was an "existing mine" when the Regulations came into effect in 1977. Even if the MMLER were in effect legally, they would not have been effective at controlling the effects of this operation because:

1) the MMLER only apply to the quality of effluent - without a special regulation, the termination of a release of effluent as was done at the Brenda Mine, cannot be achieved with the federal regulations.

2) the MMLER only regulate effluent *metal concentrations*, not *total loading*, and the latter must be controlled if ambient water and sediment quality is to be protected;

3) the MMLER do not apply to Mo or nitrates, which were the most prevalent species in the Brenda wastewaters;

4) the MMLER do not allow for use of environmental quality objectives (such as the B.C. Water Quality Objectives which were used as a basis for prohibiting the 1980s release of surface runoff) to set allowable contaminant loading;

5) the Federal Government does not have a mandate to regulate the protection of livestock or wildlife species from the effects of Mo releases from this mine to surface water bodies; and

6) the MMLER do not apply to decommissioned operations once the active discharge of effluents ceases, although the most significant releases of effluent may be expected to

occur in the post-operational stage of the Brenda mine.

The Provincial Waste Management Act (WMA) permit, including amendments (MELP 1969, 1980, and 1994), was used to curtail the effects of minesite drainage on the quality of downstream waterbodies. The WMA allows the Regional Waste Manager to "*specify procedures or requirements respecting the handling, treatment, transportation, discharge or storage of waste that the holder of the permit must fulfill...*" (Section 8(1)(e)). Thus, the WMA permits the prohibition of discharges, the regulation of effluent quality and discharge rates to account for dilution and other assimiliatory processes, and the use of environmental quality objectives as a basis for setting effluent-release loading rates, including the protection of all water uses. Permission to discharge effluents, and restrictions on contaminant loading, may also be based on watershed management plans, which can take into account the effects from multiple waste discharges and water withdrawals, and can allow effluent trading to prevent further degradation of water quality.

3.5 Assessment of Monitoring-program Adequacy

Given the need for a positive discharge of minesite runoff, including tailings-pond and open-pit runoff, and the uncertainty in meeting the stringent criteria for Mo in irrigation-water sources (0.01 mg/L; Bradburn and Perkins 1995), it will be necessary to have a water- and sediment-quality and biological baseline that will be capable of early detection of effects. The Rescan (1993a) water-quality baseline in the creeks and downstream lakes is likely adequate as long as water-quality objectives are used as the effect indicators. However, a better defined seasonal baseline should be collected, and the data should be supported with the appropriate information on quality assurance in order to detect changes in water quality prior to exceeding the concentrations stipulated in water-quality objectives.

Assessment of the effects of metal accumulation in sediment-disposal areas in the downstream lakes and appropriate creek reaches will require better representation of surface-sediment chemistry. Sufficient baseline replicates for sediment and benthic macroinvertebrate analyses should be collected in well-defined deposition areas to enable changes to be detected before significant biological effects have occurred.

3.6 References

Bradburn, R. and Perkins, D. (1995) *Decommissioning of the Brenda Mine near Peachland, British Columbia.* Paper No. 18, Proceedings of the Canadian Mineral Processors 27th Annual Meeting, Ottawa, Ontario.

*Bryan, J.E. (1991) *Some Effects of Wastewater from Brenda Mines on Water Quality, Aquatic Organisms, and Forage Crops*. British Columbia Ministry of Environment, Penticton, British Columbia. (AQUAMIN Report No. 396).

Ministry of Environment, Lands and Parks (1969) *Brenda Mines Ltd. Permit PE 00263 under the Waste Management Act*.

*Rescan (1992a) *Chemical Diagenesis of Sediments in Peachland Lake, B.C.* Consultant report with appendices prepared for Brenda Mines Limited, Peachland, B.C. (AQUAMIN Report 354).

*Rescan (1992b) *Assessment of Rainbow Trout Populations in the Lower Reaches of Peachland and Trepanier Creeks*. Consultant report with appendices prepared for Brenda Mines Limited, Peachland, B.C. (AQUAMIN Report 356).

*Rescan (1993a) *Brenda Mine Decommissioning Plan, Volume II, Environmental Baseline Studies*. Consultant report prepared for Brenda Mines Limited, Peachland, B.C. (AQUAMIN Report 374).

*Rescan (1993b) *Okanagan Baseline Study Benthic Invertebrate Assessment*. Consultant report with appendices prepared for Brenda Mines Limited, Peachland, B.C. (AQUAMIN Report 355).

Soregaroli, A. E. and Whitford, D. F. (1976) Brenda. *CIM Special Vol.* **15**, 186-194.

***AQUAMIN Reports**

354: Rescan (1992a)
355: Rescan (1993b)
356: Rescan (1992b)
374: Rescan (1993a)
396: Bryan (1991)

4 Myra Falls Mine

4.1 Introduction

The Myra Falls flotation mill and underground mine on Vancouver Island, British Columbia, have produced Zn, Cu, Ag, Pb, and Au in concentrates since the start of operations in 1967. Although the operations met the standards of mining and legislation during the 1960s and 1970s, the receiving environment degraded. Once the cause was found, the mine installed a water-collection and treatment system, which resulted in recovery of the aquatic environment of Buttle Lake. Because of its proactive stance and environmental awareness, the mine has continued to complete studies and upgrade treatment systems to ensure environmental protection as mine operations proceed.

The *Metal Mining Liquid Effluent Regulations* (MMLER) were not introduced until ten years after mining began. These regulations did not effect environmental protection. Environmental recovery was possible as a result of the analysis of monitoring data from the receiving environment; however, although it conformed with previous legislative requirements, the monitoring program was not as effective as it should have been. The following sections describe the Myra Falls mine operations, aquatic environmental effects, and the recommendations for legislation and environmental monitoring that would have provided better environmental protection.

4.2 General Description of the Minesite

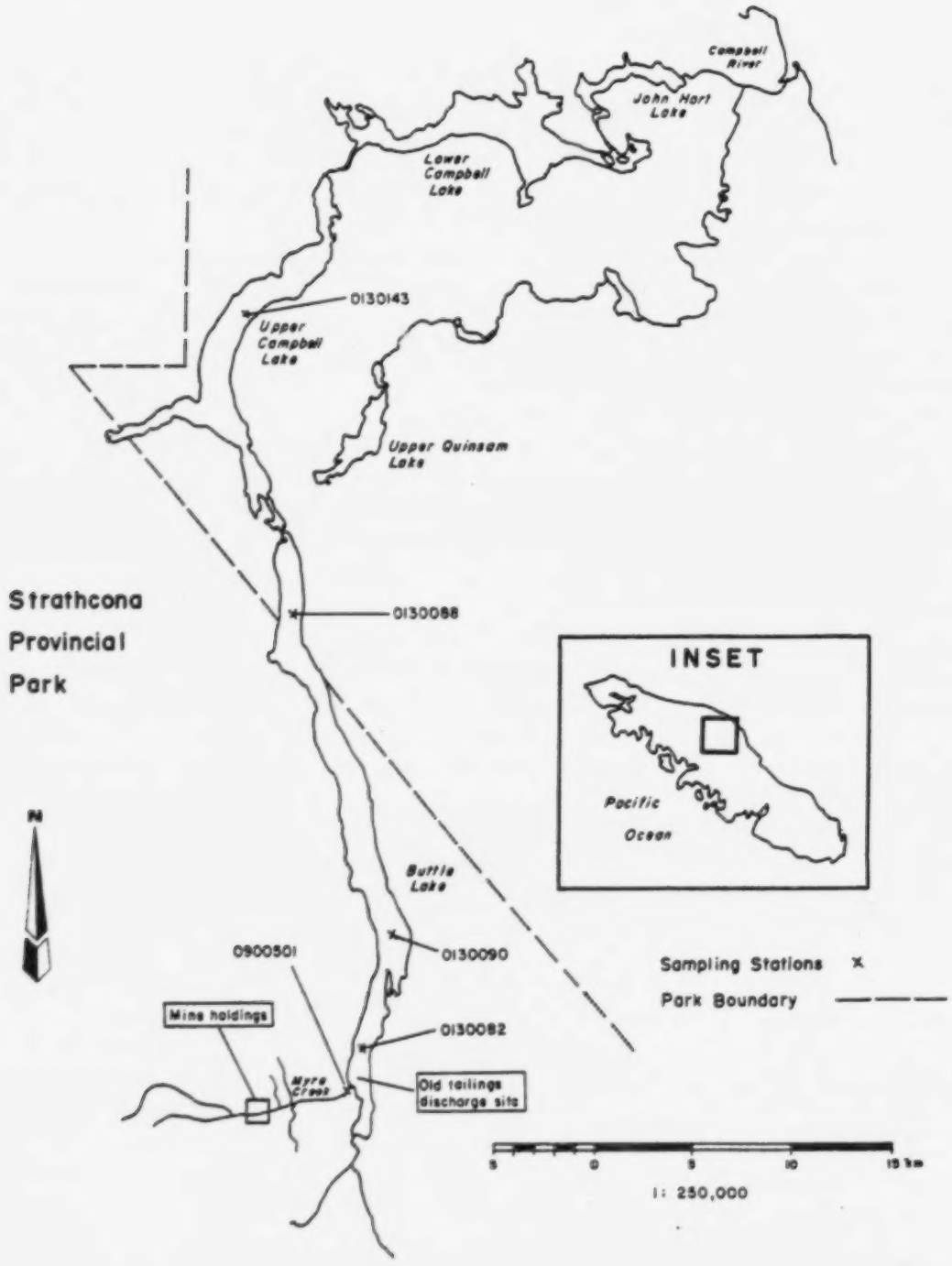
a) Location

The Myra Falls Study Area is located at the South end of Buttle lake in Strathcona Park, approximately 93 km from Campbell River, on Vancouver Island, B.C. (Figure 4.1).

b) Geology

The ore deposits are complexly metal zoned volcanogenic massive sulphides of Devonian Age, hosted in the Myra Formation of the Sicker Group volcanic assemblage. Three distinct rhyolite horizons are defined within the 450 metre thick Myra Formation and massive sulphide mineralization is associated with each. There are a number of orebody types including polymetallic massive sulphides, polymetallic disseminated sulphides, zoned pyritic

Figure 4.1: Location of the Myra Falls Mine



massive sulphides, clastic sulphide zones and stringer zones. Individual orebodies range in ize from 10,000 tonne zinc rich polymetallic lenses to the 10 million tonne, zoned massive sulphide H-W main lens. The Lynx and Myra deposits lie at the same stratigraphic level of the Myra Formation (the Mine Sequence of the Juras). The H-W horizon lies below them at the base of the Myra Formation.

The lenses of massive sulphides at the Myra and Lynx mines occur in a gangue of quartz sericite talc and chlorite with lenses up to 12 metres thick and 24 metres long (GSC Minfile#092071 p.7 #092072 rev,1990 p.3).

Those of the H-W mine are up to 30 metres thick pinching out along strike with an average thickness of 18 metres; here the mineralization is strongly aligned laterally with a massive pyrite core containing high copper-zinc to zinc ratios with barite rich margins containing low copper to zinc ratios (GSC Minfile#092F330 rev.1990 p.17). The pyrite content of the H-W deposits is much higher than other deposits in the area and averages 70% in the ore grade of the mine (Ibid p.17). The lenses of massive sulphides comprise banded chalcopyrite ($CuFeS_2$), galena (PbS), sphalerite ((Zn,Fe)S), barite ($BaSO_4$), and pyrite (FeS_2) along with other less significant minerals. It is from these sulphide deposits that most of the commodities mined in this operation come from: copper, lead, zinc, silver, and gold.

The sulphide mineralization of this area is known to be conducive to oxidation and acid generation. Pyrite exposed to oxygen (air) and water oxidizes to sulphuric acid and ferric iron. The acidity and ferric iron solubilize sulphide minerals such as chalcopyrite and sphalerite and mobilize contained metallic elements (notably copper and zinc). Where pyrite oxidation proceeds in a flowing stream of water, such as precipitation runoff, the stream is contaminated with dissolved metals. This process occurs in nature, but is enhanced by mining when rock fragmentation exposes more pyrite to oxidation than would be available naturally.

c) General History of Minesite Operations

At the time of establishment of Strathcona as a park in 1911 the boundary included mineral rights already held by Western Mines (the predecessor of Westmin Resources Limited). In 1965 a Parks Use Permit (No. 477) was granted to Western Mines which gave them surface rights to allow for the construction of a mine-mill complex (History of Western Mines with respect to Pollution Control Act Clark report 1980).

Westmin Resources Limited has operated both open pit and underground mines at Myra Falls. The mining was originally carried out by cut and fill and blasthole stoping methods which were later switched to a mechanized room and pillar with a variation on the Avoca

method of sublevel retreat (CIMM 87th AGM).

The combined milled daily production of the H-W Lynx and Myra mines in 1967 started at 875 tonnes; this increased to 2700 tonnes with the expansion to include the H-W mine in 1985. 1993 capacity has declined slightly from the 1988 figure of 4000 tonnes per day to 3,386 tonnes per day. (GSC Minfile# 092F071 p3 Westmin 1993 A.G.R.p.5)(Figure 4.2).

Currently the H-W mine is in operation; the Myra and Lynx mines have closed; the Battle and Gap deposits with the second and third largest mineable proven and probable reserves (combined total of 3,571,300 tonnes) along with other less significant deposits have yet to be used as mill sources (although the Battle Zone deposit is slated for production in the near future) (Westmin 1993 A.G.R.p.7).

The method of milling is by conventional Differential Flotation; 1992 additions included two gravity concentrators to facilitate high grade gold extraction.

Initially an alkaline chlorination treatment system was established in December 1973 for cyanide destruction. A chlorine evaporator and chlorine filter were later added to the treatment system (1975, 1976, respectively) to remove impurities from the feed (CIMM 86th AGM Apr./84). Treated effluent from the plant and tailing fines were then deep-disposed to Buttle Lake. This practice was discontinued in July 1984. The alkaline chlorination treatment system was removed in 1985 after the site stopped using sodium cyanide as a flotation conditioner.

As of July 1992 treated waters were combined with Lynx mine water and disposed to the Lynx settlement pond (to further facilitate metal removal) (Westmin Water Management Plan July 1992). Mine water from the Lynx and Myra mines was originally directed to settling ponds prior to discharge to Myra Creek. In 1982 and 1983 additional systems were added to collect waste dump runoff and seepages and additional ponds were created treated with lime and settled before the waste water was released to Myra Creek (Figure 4.3) (CIMM 86th AGM 1989).

d) History of Specific Mines and Related Effluent Quality

First production took place at the no longer operating Lynx open pit mine in 1967. In 1969 occasional non-compliance was noticed relating to dissolved copper and zinc (Pollution Control Branch Cu 0.02 ppm and Zn 0.8 ppm). The 1971 monitoring results showed the mine was out of compliance for cyanide (Pollution Control Branch #11 Vol.2). In 1975 the Lynx open pit closed.

Figure 4.2: Myra Falls timeline

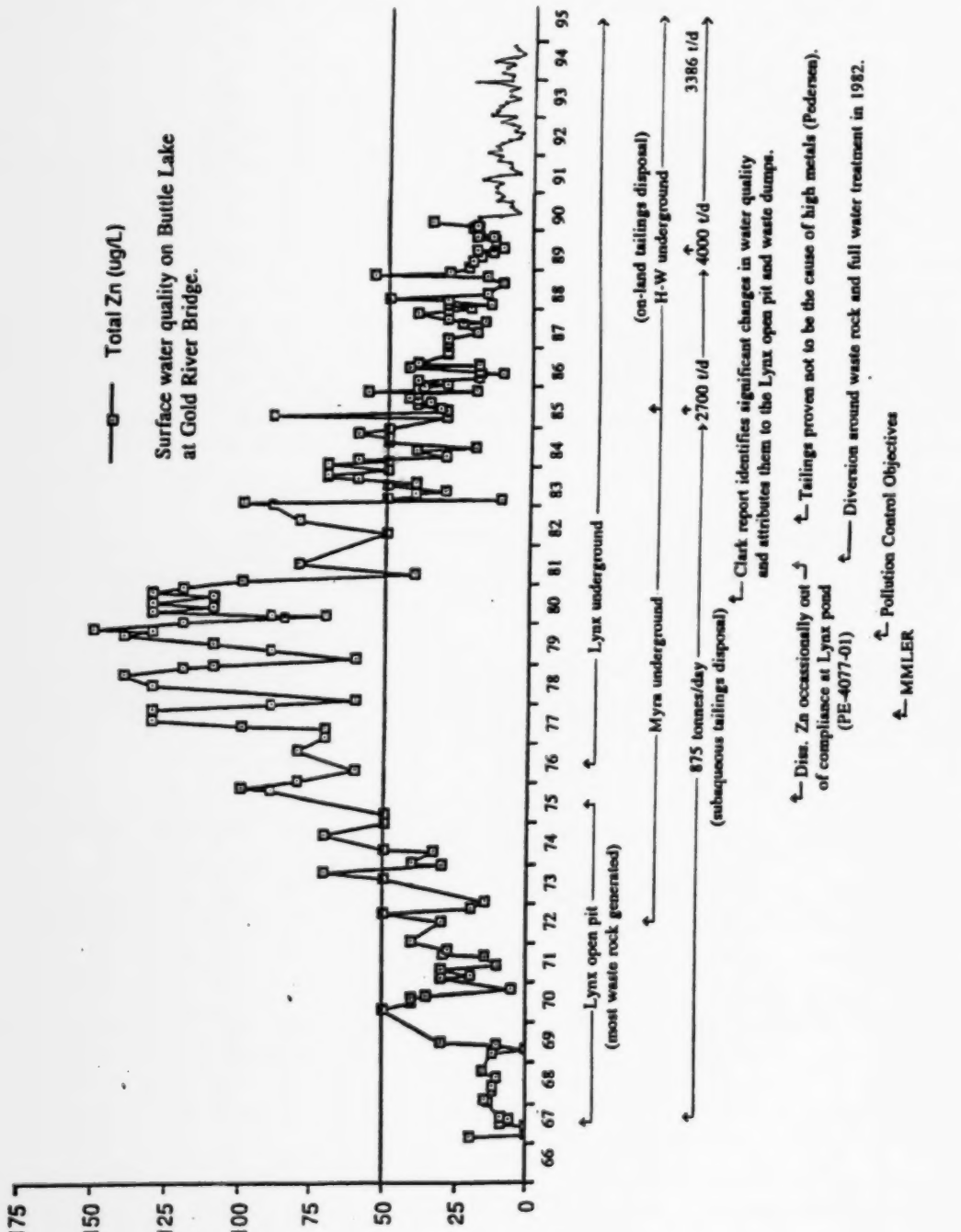
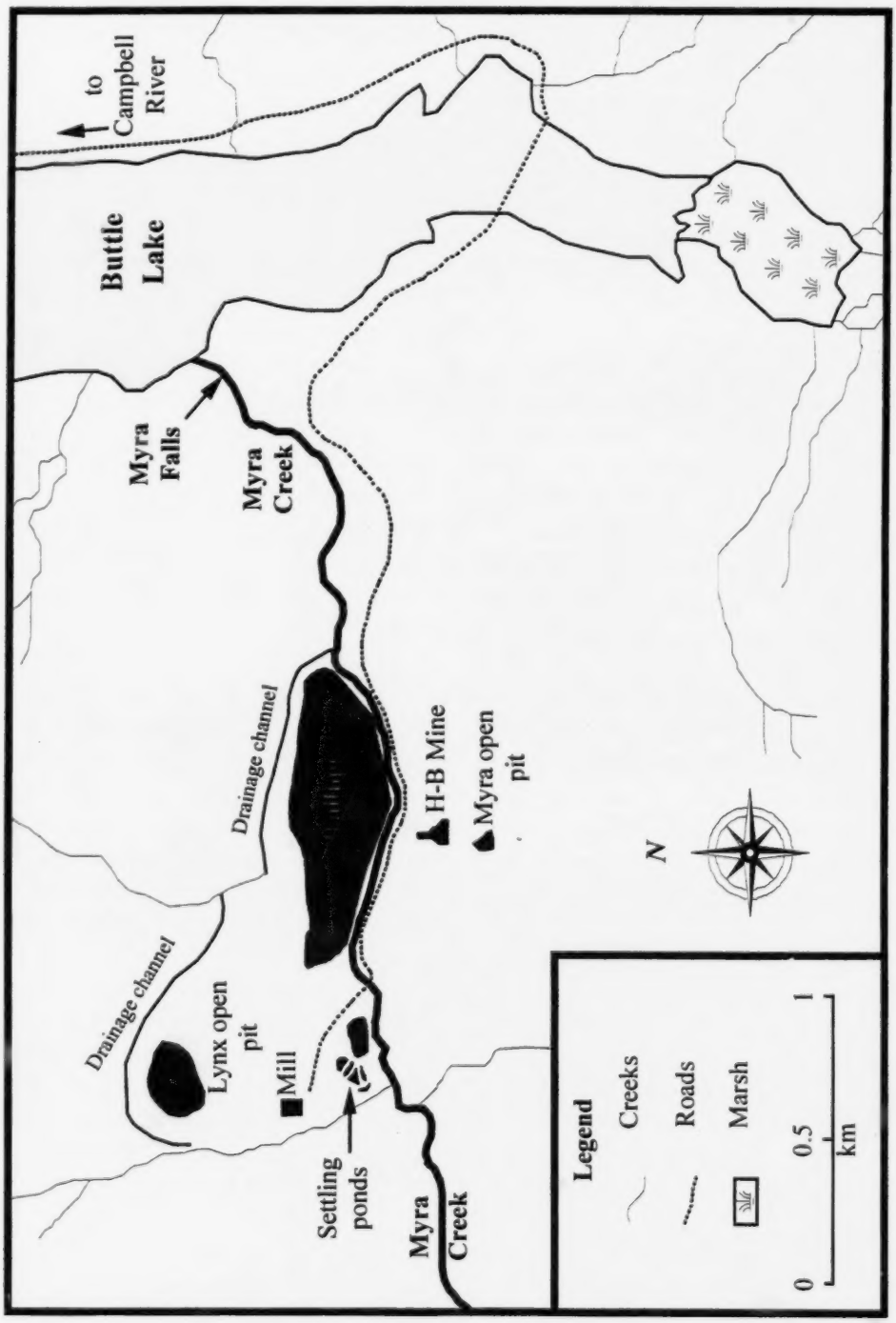

Total Zn (ug/L)
Surface water quality on Buttle Lake at Gold River Bridge.
175
150
125
100
75
50
25
0
66 67 68 69 70 71 72 73 74 75 76 77 78 79 80 81 82 83 84 85 86 87 88 89 90 91 92 93 94 95
Lynx open pit
(most waste rock generated)
Lynx underground
Myra underground
(on-land tailings disposal)
H-W underground
875 tonnes/day
(subaqueous tailings disposal)
2700 t/d
4000 t/d
3386 t/d
Clark report identifies significant changes in water quality and attributes them to the Lynx open pit and waste dumps.
Diss. Zn occassionally out of compliance at Lynx pond (PE-4077-01)
Tailings proven not to be the cause of high metals (Pedersen).
Diversion around waste rock and full water treatment in 1982.
Pollution Control Objectives
MMLER

Figure 4.3: Site plan for the Myra Falls Mine



In 1972 the Myra underground mine (latitude: 49 34 08N and longitude: 125 35 31W) came on line and the overall grade of total ore milled here was 2.16 grams per tonne gold 81.0 grams per tonne silver 1.83 % copper 0.78 % lead and 6.58% zinc (GSC Minfile #092F 072 ev.1990 p.7) until its mineable reserves were depleted in 1985. The Myra drainage pond treats mine drainage from the Myra mine. All water exfiltrates and joins other surface and ground waters prior to being sampled. Between 1977 and 1980 the Myra ponds showed occasional non-compliance with cadmium levels from 0.0051 to 0.05 mg/L. In 1979 dissolved copper levels were in the 0.31 to 0.45 mg/L range.

From 1976 to 1993 the Lynx underground mine (latitude: 49 34 04N and longitude: 125 36 13W) was in operation. The overall grade of the ore milled here was 2.16 grams per tonne gold 81.0 grams per tonne silver 1.83% copper 0.78% lead 6.58% zinc (GSC Minfile #092F 071 p.3) Proven and probable geological reserve ore grades (with the exception of copper) are somewhat higher at 3.1 grams per tonne gold 95.8 grams per tonne silver 1.1% lead. Up to the end of 1988 the Lynx mine produced over half of a total of 9170 609 tonnes milled at the Myra Falls operation falling to 42815 tonnes when it closed in 1993 (Westmin A.G.R 1993). Mine drainage from the Lynx operation and mill was directed to a series of ponds before discharge into Myra Creek.

From 1975 to 1981 leachate from the Lynx open pit and from the settling ponds showed occasional non-compliance with dissolved cadmium and zinc levels of 0.0076 to 0.05 mg/L and 10.1 to 50.0 mg/L respectively (Clark M.J.R. 1980).

The federal MMLER's were legislated in 1977. New permits were written for the H-W mine expansion in 1985 but were not affected by MMLER (Erickson pers. comm. 1995). Provincial Pollution Control Regulations came into effect in 1979.

Elevated metal levels in Buttle Lake were of concern around 1980 and were identified as coming from the waste rock dumps and exposed walls of the open pit (A preliminary review of Buttle Lake water quality MOE Report No.80-2.#137).

In response to the results of the Clark report (1980) on zinc and copper leaching from the Lynx open pit and pyritic waste rock dump Westmin installed a fully integrated water treatment system (consisting of lime addition and five settling ponds) in August 1982 to collect and treat subsurface and surface discharge into the creek; diversion of surface runoff around the waste rock also took place.

After 1981 piezometers were installed and groundwater was monitored on a regular basis.

The December 7 1982 monitoring results on Myra Creek downstream of the mine still showed elevated metal loadings which were found to have originated mainly from the

tailings pipeline road (constructed of waste rock).

The 1983 monitoring results showed some elevated copper and zinc loadings below the Lynx pond discharge.

Site drainage or pond effluent was said to have accounted for accounted for 63% of the dissolved copper loading 74% of the total zinc loadings 92% of the dissolved zinc loadings seen below the Lynx pond discharge (EC Progress Report #2 p.43 AquaDB#582). In 1984 the mine tailings which had been traditionally piped to Buttle Lake for disposal were then diverted to an on-land disposal system combined with a seepage collection and treatment system.

Expansion to include the H-W mine (latitude: 49 34 24N and longitude: 125 35 25W) took place in 1985.

As of 1993 the H-W mine operation was responsible for 90.4% of the total ore milled at the Myra Falls operation. The grade of proven and probable mineable reserves of the H-W deposit is 1.9 grams per tonne gold 33.0 grams per tonne silver 1.50% copper 0.40% lead and 3.8% zinc (Westmin AGR 1993).

From 1985 to 1990 bi-monthly and quarterly monitoring results at Station 0900501 in Myra Creek showed occasional fluctuations of non-compliance relating to total copper and total zinc.These appear to be related primarily to annual flow variations (MoE Deniseger J. and Erickson L. 1991).

The most recent Effluent Regulations for the discharge of minesite effluent were issued on June 11 1990 and parameters were set under Permit PE6858 as follows :

Parameter	PE6858	MMLER
Total Suspended Solids	25 mg/l	50 mg/l
pH	<11	5.0 minimum
Total Cu	0.6 mg/l	0.6 mg/l
Dissolved Cu	0.2 mg/l	NA
Total Zn	1.0 mg/l	1.0 mg/l
Dissolved Zn	0.5 mg/l	NA
Dissolved Cd	0.005 mg/l	NA
Dissolved Pb	0.05 mg/l	NA
Toxicity	100% (96hr LC_{50})	50% (96hr flow through)

In addition, a number of receiving environment parameters are to be tested in Myra Creek and Buttle Lake. 1990 to 1993 daily monitoring results undertaken by Westmin from the Myra Falls Water Treatment System show the quality of effluent has improved significantly showing complete compliance for dissolved lead and cadmium and total copper results in 1993. Toxicity is also shown to be in compliance at this time. As of 1993 total zinc and pH were slightly out of compliance (by 0.3% and 7.1% of the time respectively).

Metal levels in Buttle Lake are now in compliance with the Provincial water quality standards for the protection of aquatic life with the exception of Cu (5 μg/L) which could be a result of natural sources. Westmin's water quality monitoring results in Myra Creek still show moderately elevated levels of metal loadings (Westmin Resources A.G.R.1993). In addition, Westmin has added a Superpond to the treatment system which increases the storage capacity to manage storm events and associated elevated total zinc concentrations.

e) Other Site Information

In general there are no other disturbances affecting Buttle Lake in the vicinity of the mine operations. Further down the watershed there has been timber harvesting the establishment of a hydro electric power project, a coal mining operation (Quinsam), and some human settlement.

4.3 Assessment of Information Concerning Environmental Effects

a) Available Information

Available environmental information for the Myra Falls operation consists of 19 reports listed in the AQUAMIN database. Of these 19 reports, 7 were reviewed during the first round of effects analyses. In these reports (reports 404, 10, 11, 12, 416, 215, and 26) analyses of water chemistry, zooplankton, phytoplankton, macro- invertebrates, and fish-tissue samples were deemed to be of suitable quality for determining effects on the basis that they met the criteria for quality control, reference sites, and data analysis. To identify the cause/effect relationships, a more comprehensive analysis of relationships between combinations of abiotic and biotic factors was required. Sediment data recorded in Report 215 consist of baseline information on tailings; although effects are not identified in the report, the information can be used in conjunction with that in other reports.

Following the first-round review, 12 reports (579, 578, 582, 508, 526, 3, 119, 496, 580, 729, 137, and 593) were added to the list to review. These reports included water-quality monitoring conducted during 1982, 1983, and 1987, a fish-tissue metallothionein study, a

study of the Campbell River estuary, a preliminary statistical evaluation of Buttle Lake water quality, and a few serial papers on periphyton, phytoplankton, and the biota surrounding the minesite. In view of the net gain of information, these 12 reports were not individually reviewed, but were used in the overall case study. Information was used only if it had adequate quality control. These reports were also utilized in assessing the effectiveness of the monitoring program, and in determining data gaps. To remedy information gaps, the reports listed above were supplemented with other published reports, government compliance data, permits, monitoring data, and the reviewer's general knowledge of the project.

b) Information Summary

Information predating mine construction relates to studies carried out in conjunction with hydroelectric development on the Campbell River, and includes fisheries studies completed primarily in the lower Campbell River. A study of the fish in Buttle Lake was completed in 1953, prior to the increased flooding of the lake in 1955 which resulted in a 26% loss of Buttle Lake spawning habitat from the tributary creeks. Initial phytoplankton studies were completed in 1966 and 1967, and were published in 1981 along with a 1980 phytoplankton survey of Buttle Lake.

At the start of mining in 1967, studies were completed by B.C. Research Council and the Campbell River District Pollution Control Society to monitor the impact of tailings discharge on Buttle Lake. At that time the mine was required to complete quarterly water-quality sampling of Buttle Lake and an annual survey of fish-tissue metals. This program was expanded in 1974 to three fish collections per year. Bioassay toxicity testing of tailings effluent was required twice annually beginning in 1975.

Numerous studies relating to water quality, limnology, and fisheries were carried out from the start of mining in 1967 and continued into the late 1970s. A report by government agencies (Clark 1980) completed a statistical analysis on Buttle Lake water quality which was the first report since mining began that revealed a statistically detectable increase in Zn concentrations. Additional studies were completed in the early 1980s as a result of the recognition of the increased metal levels in Buttle Lake. These studies were followed by a number of others implemented in the early to mid-1980s to deal with the expanded exploitation of the H-W orebody, the collection and treatment of acid rock drainage, and the construction of a surface tailings-disposal system.

The majority of historical data on the biology of Myra Creek is contained in the Stage II Report (Westmin Resources Limited 1982) prepared for the H-W Expansion Project. These data, which include the following, were gathered before the H-W Expansion was completed; the data also precede the diversion of Myra Creek and the initiation of water collection and

treatment:

- fisheries biophysical and habitat inventory from the Tennent powerhouse to Myra Falls
- fisheries species and population estimates
- fish characteristics (size and age)
- benthic macroinvertebrates (identification to order only)

Additionally, a periphyton study was completed in 1986, and water-quality monitoring has been carried out on Myra Creek continuously since the H-W Expansion Project was completed.

Bi-monthly sampling of the water quality of Buttle Lake began in 1983, at which time the responsibility for monitoring was transferred from the government to the mine. Biological parameters were not sampled until 1989. At present, the following monitoring program is required (under permit PE 6858 and previously under permit PE 6582):

- quarterly sampling of tailings supernatant (pH, total suspended solids, total N, total P, and total and dissolved Zn, Cu, Cd, and Pb)
- continuous recording of pH from the Myra Ponds in addition to daily, monthly, or quarterly measurements of the other parameters measured for tailings supernatant
- quarterly 96-h LC_{50} toxicity testing of the Myra Ponds
- monthly monitoring of pH, and total Zn and total Cu for the Price Ponds
- quarterly water quality on Myra Creek site M-1, monthly sampling of water quality on Myra Creek site M-2, and below "Pumphouse Report 4"
- daily flow rates on Myra Creek
- monthly water chemistry at 10- to 20-m depth intervals on Buttle Lake at two sites
- monthly water-quality profiles at two sites on Buttle Lake (north and south basin) for temperature, dissolved oxygen, pH, and conductivity
- periphyton and zooplankton cell counts, species identification, and chlorophyll-*a* at the same two Buttle Lake locations
- water chemistry at surface in Buttle Lake at the Gold River bridge

In addition, studies of the Buttle Lake tailings have been completed by T. Pedersen at the University of British Columbia, and in conjunction with MEND. Although these reports were not submitted for review, the results indicate that the tailings liberate very little, if any, metal to the overlying water column, and that the tailings have low reactivity with the surrounding waters (Pedersen, personal comm. 1995).

c) Data Quality and Data Gaps

In hindsight, the regulatory requirements for environmental monitoring and independent environmental studies completed during mine operations from 1967 to 1985 had shortcomings which included:

- assigned sample stations were broad-ranging prior to 1983 and were not capable of identifying the specific source of contaminants which caused the change in Buttle Lake water quality
- there was no requirement for biological monitoring of Myra Creek
- continuous feedback and ongoing data analysis have not been frequent enough for the mine to respond effectively
- water-quality sampling equipment used by some regulatory agencies in the late 1960s and early 1970s was constructed of metal, thereby potentially introducing contaminants, which made the integrity of the data questionable
- measured parameters have not been consistent among regulatory agencies, making impossible the identification of changes in water quality
- inconsistent periods of monitoring made the determination of differences between sites difficult to separate from natural annual variation
- baseline data were not required by government agencies, and therefore were limited to regional studies originating with the Campbell River hydroelectric project
- baseline mine-waste characterization tests which would determine the potential risks from acid rock drainage were not required by government agencies
- assumptions in some of the studies were made, without evidence, that the source of metals was the tailings [both Clark (1980) and Reid and McGreer (1981) allude to this assumption]

Despite these gaps, the studies and overall information collected during the life of the mine were sufficient to identify the main effects from the Myra Falls mine. Monitoring effluent under the existing regulations did not provide adequate information for environmental protection. In some cases the collected data were extraneous to the needs of the program. This situation could be avoided in the future by a greater focus on how the data will be analyzed; as well, the selection of appropriate analyses should be done during the design phase of the program, prior to initiating monitoring. Both of these methodologies should be completed in consultation with the company.

4.4 Results of Aquatic Effects Assessment

The Myra Falls waste-management permits were written in 1967, prior to the MMLER (1977). The next waste-management permits were written in 1984 as a result of the H-W

mine expansion and on-land tailings disposal. As the MMLER were not used in the writing of these permits (Erickson, personal comm. 1995), aquatic effects from the mine cannot be directly related to the effectiveness or ineffectiveness of the MMLER. However, aquatic effects did result from non-compliance in the late 1970s with the Waste Management Permits written by the B.C. Ministry of Environment and, in part, because of an ineffective monitoring program even when the mine was in compliance. The following sections discuss the measured aquatic effects.

Temporal and spatial changes in the receiving environment and aquatic biota were detected several years after the start of operation of the Myra Falls mine as discussed in the following sections. Changes were identified in water quality and fish-tissue metals only as a result of statistical analyses of historical data that were collected in the monitoring program specified by the B.C. Ministry of Environment (Clark 1980). However, the studies that analyzed the monitoring data were completed separately from the monitoring program and on an irregular time schedule. Effects on phytoplankton, zooplankton, periphyton, and metallothionein were identified by subsequent independent studies (Austin *et al.* 1985, Deniseger *et al.* 1986, Deniseger *et al.* 1990, BC Research 1974, McKean *et al.* 1990). Following these studies, phytoplankton and zooplankton sampling was added to the regular monitoring in 1989 in order to continue tracking of the improvements to the plankton communities. The aquatic effects observed at the Myra Falls mine do not relate directly to the effectiveness of the MMLER.

Aquatic effects, at least to some extent, were observed for all parameters; the distribution is related to the ecological connections in the Buttle Lake watershed. For example, release of metals from waste-rock runoff may result in a sequence of related ecological events as follows:

- increased metals in surface water and groundwater cause increased metals in stream sediments
- this in turn results in higher metals in algae (periphyton and phytoplankton), and changes in the overall composition of the algal community (some metal-intolerant species disappear and others become more dominant)
- zooplankton and benthic macroinvertebrates are affected through general assimilation of metals from the water column and ingestion of algae, resulting in a shift of species composition and potentially lower densities
- fish subsequently accumulate metals in body tissues and produce metallothionein in their livers as a result of metals assimilated from the water column, from sediments (primarily in bottom feeders), and from ingestion of zooplankton and benthic macroinvertebrates

Effects on the individual parameters over the life of the mine are presented in the following

subsections. Authors for AQUAMIN Reports 526, 508, 3, 119, 10, 11, and 12 were not cited when another more recent report contained the same information.

a) Water Quality

From the beginning of operations to 1980, levels of Zn in Buttle Lake increased with time at all sites and depths examined (Clark 1980). Prior to the implementation of the water-collection and treatment system in 1983, maximum total concentrations in Myra Creek, downstream of the minesite (Environment Canada Station 0900501), were found to be 2.31 mg/L Zn, 0.170 mg/L Cu, 0.0045 mg/L Cd, and 0.004 mg/L Pb. Many concentrations were higher in winter months, when high rainfall and rapid runoff result in greater quantities of water percolating through the waste-rock dumps. Metal levels in Myra Creek improved after the installation of the water-collection and treatment system, and during summer (1989/90) generally remained below 0.07 mg/L Zn, 0.006 mg/L Cu, and 0.004 mg/L Cd. Concentrations in winter generally remained below 0.300 mg/L Zn, 0.030 mg/L Cu, and 0.008 mg/L Cd.

At site 0130080 (Buttle Lake at Gold River bridge; 30 km downstream) concentrations in 1980/81 ranged from 0.07 to 0.15 mg/L Zn, 0.003 to 0.010 mg/L Cu, and <0.001 to 0.013 mg/L Pb. By 1989/90, concentrations had decreased and ranged from 0.01 to 0.055 mg/L Zn, <0.001 to 0.005 mg/L Cu, and <0.001 to 0.003 mg/L Pb.

At site E207772 (Campbell River, approximately 70 km downstream) 1989/90 concentrations ranged from <0.005 to 0.015 mg/L Zn, 0.001 to 0.004 mg/L Cu, <0.001 to 0.002 mg/L Pb, and <0.0001 to 0.0002 mg/L Cd (Deniseger and Erickson 1991).

Metal levels throughout Buttle Lake have improved since the early 1980s, and now meet the B.C. water-quality criteria for the protection of aquatic life; the exception is that Cu, which may be derived from natural sources, exceeds B.C. criteria of 5 μg/L throughout the year at depth, at surface in winter, and naturally in Myra Creek upstream of the mine. Metal levels in Campbell River generally meet criteria. Metal levels in Buttle Lake in 1991 showed elevated concentrations at depth and distance (Deniseger and Erickson 1991), which is expected because of the time it takes for metal loadings in the lake from previous operations to migrate downstream.

Following installation of the water-collection and treatment system, the Environmental Protection Service completed studies to assess the effectiveness of treatment. Treatment was effective in decreasing Zn loadings by 75% in lower Myra Creek, although there were still some unidentified sources of metal loadings (Kelso and Jones 1983). The source was subsequently identified as the tailings-line road, which was constructed of waste rock and

was not included in the original collection and treatment system.

Following installation of the treatment system, Environment Canada's Environmental Protection Service continued to take samples. Metal and sulphate levels were lower in May and September 1983 than in December 1982 (Ross and Jones 1983). Samples taken in September 1983 contained elevated levels of Zn in Buttle Lake at the Gold River bridge (Godin *et al.* 1985). Sampling completed in 1985 during a period of high rainfall showed high metal levels in Myra Creek Station 7, which were attributed to contaminants in the effluent from the Lynx and Myra mines, the old tailings-line road seepages, and possibly seepages from the groundwater collection system (Godin and Jones 1987). The tailings road was removed to reduce loadings to creek water in 1982.

Metal levels from 1990 to 1994 have declined to near background. The regular annual variation is due to the seasonal turnover in the lake; slightly higher metal values are observed in winter months, whereas in summer the decreased values are either at, or slightly above, current levels of detection.

b) Zooplankton

Zooplankton communities in Buttle Lake that were affected by high Zn levels in the 1970s have improved dramatically in both density and diversity since the installation of the water-treatment system. By 1990 the species composition of zooplankton seemed to have returned to that estimated to exist prior to mine development. Metal-sensitive zooplankton species are recovering, likely contributing to an improved rainbow trout fishery (Deniseger and Erickson 1991). Communities now include species from the genera *Diaptomus, Daphnia, Bosmina,* and *Cyclops* which were recorded during pre-mining conditions. In addition, a number of rotifer species from the genera *Kellicotia*, *Keratella*, *Testudinella*, *Conochilus*, and *Trichotia* have returned to approximately baseline populations (Deniseger and Erickson 1991). This improvement in the number of different species of zooplankton representatives in the food web has continued from the 1990 analysis through to 1994. This improvement, beginning in 1985, was the result of changes in the mine operations, and the changes were instigated because of the recognition of non-compliance with respect to water quality. The permits and enforcement were not effective in protecting the receiving environment.

c) Phytoplankton

As metal concentration improved, the phytoplankton community in Buttle Lake began to show signs of improvement; continuous phytoplankton blooms, dominated by *Rhizosolenia eriensis*, occurred annually from 1983 to 1985. By 1985, some metal-sensitive species of

phytoplankton and zooplankton began to reappear in the lake. Although several sensitive species of zooplankton have returned to prominence, some of the more sensitive species of phytoplankton have not yet recolonized the lake in significant numbers (Deniseger *et al.* 1990).

During the 1990 to 1994 environmental monitoring program, the number of different species of phytoplankton, as seen through a microscope at 1000× magnification, has ranged from 18 to 34 per sample. This may be compared to the 45 to 51 species present in the 1960s (Austin *et al.* 1985). With improvements in water quality, phytoplankton populations are returning to a steady-state community which has a moderately rich diversity of species. It is expected that additional species will recolonize Buttle Lake over the short term, restoring the community to its original diversity.

Species dominance has changed from the 1967 associations of *Rhizosolenia eriensis*, *Asterionella formosa*, *Tabellaria fenestrata*, *Tabellaria flocculosa*, *Peridinium*, and *Ceratium hirundinella*. Species decreased in the early 1980s, and metal-intolerant species disappeared, resulting in a phytoplankton association dominated by *Synedra acus* and *S. filiformis*. Community composition increased in diversity to 1990, at which time the community included genera such as *Dinobryon*, *Sphaerocystis*, *Elakatothrix*, *Gloeocystis*, *Cryptomonas*, *Cyclotella*, and *Crusigenia* (Deniseger and Erickson 1991). This trend in species composition continued to 1994, with dominant species including *Chrysochromulina parva*, *Pseudokephyrion sp.*, and *Melosira italica*. Metal-intolerant species of *Tabellaria fenestrata*, *T. flocculosa*, and *Asterionella formosa* have yet to appear regularly, but are expected to recolonize Buttle Lake in the near future (Deniseger and Erickson 1991).

d) Periphyton

Periphyton communities within Myra Creek downstream of the mine and waste-rock dumps were virtually devoid of Chlorophyta, whereas the upstream control population included Chlorophyta species and had a larger number of taxa (Deniseger *et al.* 1986). No further studies were published after this date.

e) Benthic Macroinvertebrates

Littoral (*i.e.*, sunlight penetration is to the substrate) benthic samples were taken in 1981 for Buttle, Lower Campbell, John Hart, and Upper Quinsam lakes. The highest total and average numbers of benthic organisms were found in Lower Campbell and John Hart lakes, and the lowest numbers were found in Buttle Lake. A number of pollution-tolerant species, including *Procladius*, *Thienemannimyia*, and aberrations in *Rhyacodrilus montana*, were present in the

south Buttle Lake basin. In summary, the benthic fauna in the south basin of Buttle Lake had low densities and contained several tolerant species (Kathman 1982). No further studies were published after this date.

f) Fish

Fish-tissue data for Buttle Lake prior to 1980 were limited to copper-in-liver data from 1966 and lead-in-liver data from the late 1960s and early 1970s. The 1966 copper-in-liver values for Buttle Lake fish were lower than for other lakes on Vancouver Island. The Pb content of fish liver increased between 1969 and 1971, and between 1971 and 1972 (BC Research 1974).

In 1980 and 1981, toxicity projections based on biological literature for three species of salmonids in the lower Campbell River indicated that the waters would be toxic to some juvenile life-stages of salmonids, but no actual tests were completed (Alderdice and McLean 1982). This publication is commonly misrepresented and does not actually verify the toxicity predictions.

Laboratory testing showed that tissue concentrations for Zn, Cu, and Pb increased in sculpins exposed to 100% tailings for 28 days. This study also included a comparison of metal bioaccumulation in resident biota. Levels of all metals in biota resident in lakes downstream of the mine discharge were generally higher than in biota from a nearby lake unaffected by the mine (Reid and McGreer 1981).

In the early 1980s, increased metal concentrations in lake water were accompanied by increased metal concentrations in salmonid muscle and liver tissues, and by elevated levels of hepatic metallothionein. Metal concentrations decreased throughout the mid-1980s; by late 1985, following the installation of the water-collection and treatment system in 1983, the metal levels in rainbow trout muscle tissue had improved, and hepatic metallothionein levels had also declined to significantly lower levels. Only Cu and Cd levels in liver tissue remained significantly higher than for control lakes.

Metallothionein concentrations in rainbow trout declined from 1345 ± 115 nmoles/g (dry weight) in 1981 to 320 ± 110 nmoles/g (dry weight) in 1985 (McKean *et al.* 1990). Since 1986, hepatic metallothionein (a measure of acclimation in fish) has been reduced to levels thought to be representative of minimal biological stress; the reduction correlates with decreasing metal levels in the lake. Metals in rainbow trout muscle tissue have decreased to background levels, and Cd in liver tissue has decreased significantly; only Cu remains somewhat elevated in liver tissue. The improved health of the fisheries resource is further illustrated by dramatic improvement in angler success and catch statistics (Deniseger and

Erickson 1991).

4.5 Discussion

Zinc, copper, and sulphate levels in surface waters changed significantly over the life of the mine and is the best illustration of cause and effect. Figure 4.2 (Myra Falls timeline) shows the direct relationship between mine operations and levels of total Zn in Buttle Lake at the Gold River bridge. The oxidation of pyrite and the release of acid rock drainage from waste rock at the minesite brought about a gradual increase in concentrations of total Zn in the receiving environment; concentrations peaked in 1980, at which time the Clark (1980) report identified statistically detectable changes in water quality. Immediate action was taken by diverting water around the waste-rock dump and installing the water-collection and treatment system, resulting in decreasing levels of total Zn. Residual Zn that was accummulated in the deep lake-waters during metal releases from the mine from 1967 to 1980 has been declining; the period of improving water quality (1980 to 1993) is proportional to, and reflective of, the period of increase.

Overall, water quality and biological parameters were negatively affected between the start of mining and 1980, and in subsequent years the environmental parameters began to improve. The improvement coincided with remedial actions, including the construction of the surface runoff diversion and a water-treatment system in 1982. Causes of the observed effects in the Buttle Lake watershed can generally be attributed to the mine because the studies focussed primarily on the minesite, and because no other industries, except hydroelectric development, are present at Buttle Lake. Most of the metal loadings to Myra Creek were attributable to acid generation from the natural oxidation of pyrite which is contained within the waste-rock dumps, exposed open pit, and tailings-pipeline road; the acid in turn reacted to release the metals which caused the high metal loadings in lake waters and fish, and the changes in plankton communities. Because the MMLER and early mining permits focussed only on end-of-pipe effluent sources, non-point sources of metal loadings, as well as the receiving environment, were missed completely.

Too few studies have been completed for Myra Creek to determine overall effects, but the results of these studies show that some changes in the biota of Myra Creek have occurred even during compliance with the waste-management permit. The Myra Falls timeline (Figure 4.2) gives a good indication of the problems associated with a monitoring program that provides inadequate feedback. In the initial years of operations, four agencies were monitoring Zn concentrations, and the limits specified in the Waste Management Permit had been exceeded. Because of the lack of enforcement, however, it was not determined until 1980 that Zn levels were increasing, and this was 8 to 10 years after the levels had begun to increase. Remediation action taken upon recognition of the problem led to a reduction in the

Zn levels immediately. Up to 1982, in some papers and reports it was clearly assumed that the tailings were the source of metals, although data to support this link were never presented. As such, the monitoring program was neither adequate to provide timely feedback and response, nor was it adequate to determine the source of contamination. The permit levels, although determined prior to the MMLER, were not significantly different. In fact, the amended waste-management permit isssued in 1990 contained limits slightly more stringent than those of the MMLER: total suspended sediments in effluent were required to be 20 mg/L rather than 25 mg/L under MMLER, and total Cu and total Zn were the same at 0.30 mg/L and 0.50 mg/L, respectively; as well, limits not included in the MMLER were set for dissolved Cu, Zn, Cd, and Pb. As the MMLER requirements pertaining to the collection sites have not been amended, the source of the metal loadings at Myra Falls would not have identified, and the receiving environment would not have been protected.

a) Legislation Recommendations

This case study does not help in evaluating the effectiveness or ineffectiveness of the MMLER because the impacts from mining operations, as documented in the foregoing text, originated from non-point sources of pollution, and occurred prior to the implementation of the MMLER (1977) and B.C. Pollution Control Objectives (1979). Although new permits were issued for the H-W Expansion after the MMLER and Pollution Control Objectives were legislated, changes to mine operations were a direct response both to the observed effects on receiving-water quality and to the monitored biotic and abiotic effects. Although water quality was out of compliance from as early as 1975 under the issued permits, it was not until 1980 that the problems were identified and operations were changed. The legislation in the late 1970s did not protect the environment because there was no reporting of results, and there was no enforcement requiring change when the mine was out of compliance. The prosecution in 1985 (Cox and MacLatchy 1988) took place too late for effective enforcement. During the period of the current permits, the environment has been recovering from previous impacts; thus, it cannot be determined definitively whether these permits protect the receiving environment.

Some of the cause for the poor response to changes in water quality was the legislation during this period. Permit levels were put in place for all discharges, but the monitoring was carried out by government agencies. The company did not have the responsibility for monitoring environmental changes caused by the mine until this onus was transferred to the company in 1989. For the Myra Falls operation, legislation should have included the following items:

- permit limits based on protection of aquatic life in receiving waters
- waste characterization completed prior to design and construction

- an adequate monitoring program carried out with adequate, continuous review, and feedback from government to the company

At present, all of the sources of effluent have been identified, and the Waste Management Permit 1990 amendments require receiving-environment monitoring at appropriate locations.

b) Suggested Monitoring-program Design

The design of a monitoring program is crucial to the success or failure of the program. As was observed in the case of Myra Falls, even with a fairly large quantity of data it was difficult to determine effects during the operations, and the program was not designed to identify the specific source of contaminants. It would have been advantageous in this case to have had one monitoring program, complete with a number of reviewers to ensure that the program was secure and that a continuous tracking system was in place. Duplicates should have been used to provide quality control. This would have provided the best use of resources and allowed for a reproducible and meaningful database. Such a program would be relatively expensive at the beginning, but would save money in the long run by reducing unnecessary duplication and by providing immediate response, the latter reducing the chances of a large remediation bill.

Parameters selected for monitoring should target the expected effect, and should include a number of other parameters which potentially may be affected. This selection may be achieved by completing a modified risk assessment. For the Myra Falls operation, there were four sources of contamination: site runoff, underground minewater (settling pond) discharges, groundwater, and tailings discharges. In addition to water-quality monitoring for indicator metals (e.g., Zn, Cu, Pb, and sulphates, for Myra Falls), a primary biological producer such as periphyton or phytoplankton is generally a good indicator of environmental change. Phytoplankton was identified as one of the earliest species to bioaccumulate metals and show community changes. The frequency and location of biological monitoring are critical. Because of their high natural seasonal variability, it would be advantageous to sample biological populations annually during summer growth and/or when maximum metal bioaccumulation occurs.

Government feedback on the environmental monitoring program should be provided to the company regularly, and in terms that can be understood by the company. The company should also be provided with, or should maintain, a system by which it may self-monitor the environmental data. This arrangement would allow the company to respond in a timely manner to change, and to provide treatment or to modify the monitoring program as required.

Monitoring programs should not be too onerous. A database that monitors a large number

of parameters can become unwieldy for assessments on a regular basis if not managed well. A well-designed program with a few, well-chosen parameters presented in an easily viewed database can provide an effective monitoring tool and can permit a timely response to potential problems.

4.6 Conclusions

The three objectives addressed in this case study were:

- identify the past and present effects of the Myra Falls mine on the aquatic environment
- determine the effectiveness of MMLER in protecting the aquatic environment
- identify data gaps and provide recommendations for an effective Environmental Effects Monitoring program

Waste rock from the Myra Falls mine was determined to be the main source of metal loadings to Myra Creek and Buttle Lake. These sources caused concentrations of metals to rise in surface water, resulting in decreases in plankton densities, changes in the plankton and periphyton community structure, the presence of contamination-indicator species in benthic communities, increased metals in fish tissue, and increased hepatic metallothionein in fish. Once the system for water collection and treatment was installed in 1982, all of the previously observed aquatic effects reversed, and are now at or near pre-mining conditions.

The *Metal Mining Liquid Effluent Regulations* were not legislated until 1977, ten years after mining began. Nevertheless, the permits issued to the mine prior to 1977 required the measurement of parameters and effluents similar to those that the MMLER would have required. The permits issued in 1984 now monitor all discharges. However, as the permit requirements are still based on "end-of-pipe" measurements, it is difficult to determine whether the mine would be able to react in time to protect the receiving environment. The operational changes in effect at the mine were the result of numerous monitoring programs which identified environmental changes. Data gaps were present because of the lack of requirement for baseline data, and the lack of a consistent, regular monitoring program. However, all parameters were addressed in a reactive rather than a proactive way during some point, and in the independent studies the measurements were sufficient to be able to determine effects. A more effective monitoring program would include:

- one main monitoring program of manageable size
- design by a number of stakeholders
- adequate duplication for quality control
- an adequate feedback mechanism for immediate response to environmental changes caused by the mine.

4.7 References

*Alderdice, D.F. and McLean, W.E. (1982) A review of the potential influence of heavy metals on salmonid fishes in the Campbell River, Vancouver Island, B.C. *Canadian Technical Report of Fisheries and Aquatic Sciences, No.* **1104,** 59 p. (AQUAMIN Report 416).

*Austin, A., Deniseger, J. and Clark, M.J.R. (1985) Lake algal populations and physico-chemical changes after 14 years input of metallic mining wastes. *Water Res.* **19**, 299-308. (AQUAMIN Report 119).

*Austin, A. and Munteanu, N. (1984) Evaluation of changes in a large oligotrophic wilderness park lake exposed to mine tailing effluent for 14 years: the phytoplankton. *Environmental Pollution (Series A)* **33**, 39-62. (AQUAMIN Report 508).

*Bell, L.M. and Thompson, J.M. (1977) The Campbell River estuary: status of environmental knowledge to 1977. *Report of the Estuary Working Group, Department of Fisheries and the Environment Regional Board Pacific Region, Special Estuary Series No.* 7. (AQUAMIN Report 121).

*Broughton, L.M. and Ferguson, K.D. (1990) The geochemistry of Westmin Resources Ltd. Myra Creek mine tailings after thirty months exposure of subaerial tailings. *Environment Canada, Environmental Protection Pacific and Yukon Region, Regional Program Report* **90-04**. (AQUAMIN Report 215; Appendices 216).

*Clark, M.J.R. (1980) A preliminary review of Buttle Lake water quality. *B.C. Ministry of Environment, Waste Management Branch Report No.* **80-2**. (AQUAMIN Report 137).

*Clark, M.J.R. (1982) *Impact of the Westmin Resources Ltd. Mining Operation on Buttle Lake and the Campbell River Watershed, Volumes I, II and III.* B.C. Ministry of Environment, Victoria. (AQUAMIN Reports 10, 11, and 12).

Cox, D.O. and MacLatchy, J.E. (1988) *Volume 4, Fisheries Pollution Reports, Prosecution under the Pollution Control and Habitat Protection Provisions of the Fisheries Act*. Prepared by Environment Canada and the Department of Fisheries and Oceans.

*Deniseger, J., Austin, A. and Lucey, W.P. (1986) Periphyton communities in a pristine mountain stream above and below heavy metal mining operations. *Freshwater Biology* **16**, 209-218. (AQUAMIN Report 496).

*Deniseger, J. and Erickson, L. (1991) *Trends in Water Quality of Buttle Lake and the Campbell River - Continuing Decreases in Metal Concentrations from 1987 through 1990.* B.C. Ministry of Environment, Environmental Protection, Nanaimo, B.C. June 1991, 38 p. (AQUAMIN Report 404).

*Deniseger, J., Erickson, L.J., Austin, A., Roch, M. and Clark, M.J.R. (1990) The effects of decreasing heavy metal concentrations on the biota of Buttle Lake, Vancouver Island, British Columbia. (B.C. Ministry of Environment, May 1988). *Water Res.* **24**, 403-416. (AQUAMIN Report 526).

Erickson, L. (1995) British Columbia Ministry of Environment, Lands and Parks, Nanaimo, B.C. Personal communications regarding permitting the Myra Falls mine.

*Godin, B. and Jones, M. (1987) Progress report No. 4 - October 1985, water quality sampling in Myra Creek at Westmin Resources Ltd. mine on Vancouver Island. *Environment Canada, Conservation and Protection, Environmental Protection Service, Pacific and Yukon Region, Regional Program Report* **87-07**. (AQUAMIN Report 579).

*Godin, B., Ross, M. and Jones, M. (1985) Progress report No. 3 - September 1983, water quality sampling in Myra Creek at Westmin Resources Ltd. mine on Vancouver Island. *Environment Canada, Conservation and Protection, Environmental Protection Service, Pacific Region, Regional Program Report* **85-06**. (AQUAMIN Report 580).

Kathman, R.D. (1982) *Benthic Invertebrates of Buttle and Surrounding Lakes.* Report prepared for Aquatic Studies Branch/Waste Management Branch, British Columbia Ministry of Environment, Victoria, B.C., 36 p.

*Kelso, B.W. and Jones, M. (1983) Progress report on the December 7, 1982, water quality sampling in Myra Creek after the implementation of the groundwater and surface collection and treatment system at Westmin Resources Ltd. on Vancouver Island. *Environment Canada, Conservation and Protection, Environmental Protection Service, Pacific Region, Regional Program Report* **83-18**. (AQUAMIN Report 578).

*McKean, C.J.P., Deniseger, J. and Imber, B.E. (1990) *Metallothionein - Copper Relationships as an Indication of Chronic Stress in Rainbow Trout from Vancouver Island, British Columbia, Canada.* (AQUAMIN Report 593).

Pedersen, T. (1995) Department of Oceanography, University of British Columbia, Vancouver, B.C. Personal communications regarding recent studies of the Myra Falls, Buttle Lake tailings.

*Reid, B.J. and McGreer, E.R. (1981) Bioaccumulation of heavy metals by sculpin (*Cottus asper*) exposed to mine tailings. *Canadian Technical Report Fisheries Aquatic Sciences*, 105-121. (AQUAMIN Report 729).

*Roch, M., Nordin, R.N., Austin, A., McKean, C.J.P., Deniseger, J., Kathman, R.D., McCarter, J.A. and Clark, M.J.R. (1985) The effects of heavy metal contamination on the aquatic biota of Buttle Lake and the Campbell River drainage (Canada). *Arch. Environ. Contam. Toxicol.* **14**, 347-362. (AQUAMIN Report 3).

*Ross, M. and Jones, M. (1983) Progress report No. 2 - May 1983, water quality sampling in Myra Creek at Westmin Resources Ltd. Mine on Vancouver Island. *Department of Environment, Environmental Protection Service, Pacific Region, Regional Program Report* 84-01. (AQUAMIN Report 582).

Westmin Resources Limited (1982) Western Mines Division, Expansion Program, Myra Falls, B.C., Stage II Submission, 6 volumes.

***AQUAMIN Reports**

3: Roch *et al.* (1985)
10: Clark (1982)
11: Clark (1982)
12: Clark (1982)
119: Austin *et al.* (1985)
121: Bell & Thompson (1977)
137: Clark (1980)
215: Broughton & Ferguson (1990)
216: Broughton & Ferguson (1990)
404: Deniseger & Erickson (1991)
416: Alderdice & McLean (1982)
496: Deniseger *et al.* (1986)
508: Austin & Munteanu (1984)
526: Deniseger *et al.* (1990)
578: Kelson & Jones (1983)
579: Godin & Jones ((1987)
580: Godin *et al.* (1985)
582: Ross & Jones (1983)
593: McKean *et al.* (1990)
729: Reid & McGreer (1981)

5 Cominco Snip Mine

5.1 Modified Case Study Assessment

The Cominco Snip Mine is an interesting example of present changes in permitting requirements in British Columbia. It also presents from one entire study, an assessment of mining effects on the aquatic environment (environmental effects monitoring). Although Snip is a gold mine, cyanidization is not used in the milling process. Therefore, it is subject to the Metal Mining Liquid Effluent Regulations. In additon, it may stimulate the discussion on whether gold mines should be included in future metal mining regulations, how permitting should be addressed, and how to conduct environmental effects monitoring programs.

The Snip Mine is an underground gold mine located in northwestern British Columbia on a tributary to the Iskut-Stikine river system. Operations began in January 1991. The mine operates at 400 tonnes/day using shrinkage and cut-and-fill methods with ore processing using gravity and differential flotation. Cyanide is not used in the processing. Tailings are discharged to a storage facility which releases supernatant to Monsoon Lake feeding eventually to the Iskut River (Figure 5.1).

The Snip property is underlain by a sequence of Triassic feldspathic greywacke, siltstone and mafic tuffs. The sediments generally strike east-west and have variable dips to the north. This sequence has been intruded by feldspar porphyries, most notably the Red Bluff porphyry which extends along Bronson Creek. The main zone of mineralization, known as the Twin zone, lies within the greywackes 1000 metres to the south of the Red Bluff porphyry.

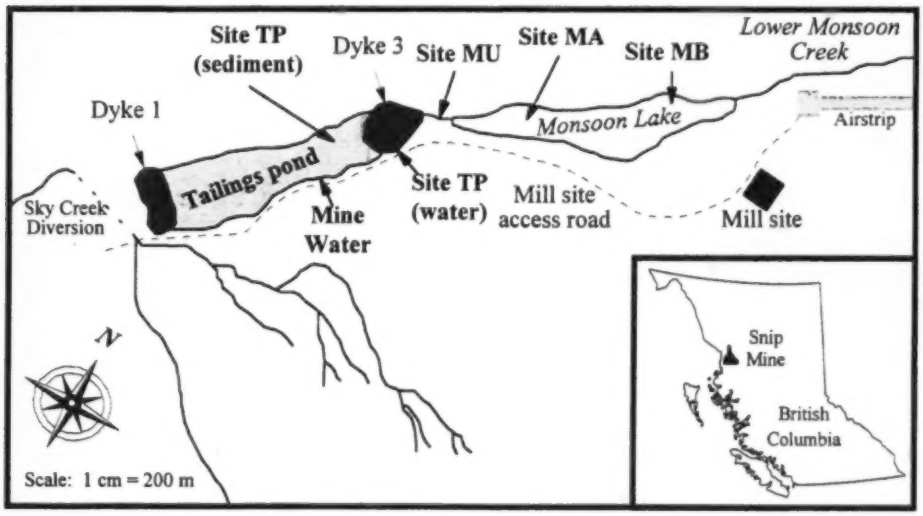
The Twin zone is a 0.5 to 15 metre wide, sheared, quartz-carbonate-sulphide vein. It has been traced by drilling over 1000 metres and has a vertical range of 500 metres. This shear hosted vein strikes at 120° and dips south-west at 30° to 90°. For most of its length the vein is divided into two parts by a post-mineralization, fine grained basic dike.

The vein contains three distinctive ore types:

1) Streaky quartz ore consisting of quartz, calcite, green biotite, and sulphide laminae within strongly sheared greywacke.

2) Crackle quarz consisting of shattered quartz vein infilled with green mica and disseminated sulphides.

Figure 5.1: Snip Mine - locations for water and sediment sample collection



3) Massive sulphide veins of mainly pyrite and pyrrhotite with minor sphalerite, chalcopyrite, and rare molybdenite, galena, and arsenopyrite.

The styles of mineralization are aligned to form a lateral zonation. The streaky and crackle quartz ores make up the bulk of the reserve in the western and central portion of the deposit. The massive sulphide ore is characteristic of the discontinuous and narrow eastern fringe.

The B.C. Waste Management permit, PE-9079 was written to incorporate a monitoring program that included not only a summary of a complete suite of tests, but required a full interpretation of the environmental impacts and recommendations for appropriate modifications to the program. The 1991 monitoring program was set up looking at the "worst case" scenario and included discussions between the company, consultant, and B.C. Ministry of Environment. Monitoring included sampling of water quality, sediment, and biota of the tailings pond and the immediate area of discharge to Monsoon Lake. Tailings pond supernatant and tailings sediments were tested for contamination and toxicity. Monthly water quality was collected from a number of locations from near the operations to downstream Monsoon Lake and Monsoon Creek. Benthic biological communities were sampled, in conjunction with sediment chemistry, and sediment toxicity tests. Acute and chronic toxicity testing of tailings pond effluent was completed over an 8-week period. Sample locations are

presented in Figure 5.1.

The additional effluent chronic and acute toxicity testing for fish, invertebrates and algae, the sediment toxicity testing and the benthic fauna community structure for the tailings pond were completed by the mine to ensure environmental protection. It was agreed that if these tests showed no adverse effects, that the subsequent monitoring would be much less stringent.

Testing was rigorous and included full quality assurance and control. The data was fully analyzed using statistical methods and interpreted.

A complete assessment was also completed in 1992 and again in 1993. 1992 studies included water quality monitoring, salmonid fish habitat assessment (including benthic macroinvertebrate sampling), 10-d sediment toxicity tests with *Daphnia*, and early life stage 7-d toxicity with salmonids. 1993 studies included water quality analysis, salmonid fish habitat assessment, and sediment quality analysis.

Permit PE-9079 effluent water chemistry limits are presented in Table 5.1 in comparison to Metal Mining Liquid Effluent Regulation Guidelines (1977) and B.C. Pollution Guidelines (1979).

The permit levels are the same as the Metal Mining Liquid Effluent Guidelines for the MMLER guideline parameters. It should also be noted that the permit addresses limits for dissolved metals, as opposed to only total metals in the federal guidelines. Tailings pond water quality for the majority of metals is well below the permitted levels.

In the tailings pond, permit levels were exceeded once for total suspended solids, dissolved aluminum, lead, and zinc, and three times for nitrite during the 1991 to 1992 study period.

Results from the 1991 testing may be summarized in four points (EVS Consultants 1992):

- Effluents and seepages did not have a negative effect on the aquatic biota in the receiving waters of Monsoon Lake. Sensitive laboratory testing on upper Monsoon Lakes sediments showed the potential for sublethal effects; however, there was no conclusive evidence based on sampling the natural populations at the same site.
- No effects were found to the aquatic organisms of Monsoon Lake.
- No effects were seen outside the tailings pond; therefore no changes to the existing permits were recommended.
- Long and short term impacts are unlikely based on all available evidence.

Table 5.1: Comparison of permit effluent limits with MMLER and B.C. Pollution Guidelines (1979).

Parameter (mg/L)	PE-9079 Limits	MMLER limits (grab sample)	B.C. Pollution Objectives	Tailings Pond WQ
TSS	50	50	25-75	12.4
Ammonia	5.0	na	1.0-10	0.75
Nitrite/Nitrate	na	na	10-25	na
Nitrite	0.2	na	na	0.115
Aluminum (dissolved)	0.3	na	0.5-1.0	0.11
Arsenic (total)	1.0	1.0	0.1-1.0	0.011
Cadmium (dissolved)	0.0008	na	0.01-1.0	0.0006
Chromium (dissolved)	0.02	na	0.05-0.3	0.008
Copper (dissolved)	0.014	na	0.05-0.3	0.007
Copper (total)	0.6	0.6	na	0.009
Iron (dissolved)	1.0	na	0.3-1.0	0.12
Lead (dissolved)	0.05	na	0.05-0.2	0.009
Lead (total)	0.4	0.4	na	0.09
Nickel (total)	1.0	1.0	na	0.045
Zinc (dissolved)	0.05	na	0.2-1.0	0.013
Zinc (total)	1.0	1.0	na	0.018

From the results of this intense study, it was recommended that the monitoring program be modified by:

- decreasing the frequency of water quality monitoring and the number of sites;
- tailings effluent toxicity testing only once a year for fish eggs;
- sediment toxicity testing for daphnids once a year for upper Monsoon Creek;
- sediment monitoring once before decommissioning;
- Sky Creek sediments testing.

Results from the 1992 and 1993 results supported the previous finding of no aquatic effects in the receiving environment with the exception of hydrocarbons in sediments localized at one site in 1992. Hydrocarbons were then tested for in 1993 and found to be from natural

organics and a site specific diesel contaminant.

Present environmental monitoring consists of water quality sampling twice a month from April to October at 4 sites, Tailings Pond, Upper Monsoon Creek, Monsoon Lake, and Lower Monsoon Creek. Sampling then is reduced to once per month from November to March for the receiving environment sites. Fish toxicity testing is no longer required and benthic studies are completed annually. In addition portal discharges at the 130, 180 and 300 level are monitored for water quality (Blundell pers. comm. 1995). The main reduction in sampling is the requirement for toxicity testing. Annual consultation between B.C. MELP and the company are scheduled to fine tune the monitoring program based on the previous annual report that compiles all the information.

In conclusion, the Cominco Snip Mine provides a good example of an effective site-specific permitting strategy that addresses not only effluent quality, but receiving environment effects as well. The closure plan will present a hydrological challenge for Monsoon lake. Currently substantial flow to the lake is provided by the effluent. Upon closure only seepage from the tailings pond will be provided. There is a concern that Monsoon lake and the present cutthroat trout population may be jeopardized.

5.2 References

Blundell, Derick Pers.Comm. 1995. Snip Mine environmental co-ordinator for Cominco.

EVS Consultants 1992. Cominco Snip Mine Environmental Effects Monitoring Program 1991 - Interpretative Summary Report. Prepared for the SNIP Operations. Project No. 3/302-05.3.